

1 Experimentally testing the response of feral cats and their prey to poison baiting

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9 10 Abstract

11 *Context* Feral cats *Felis catus* have caused the decline and extinction of many species worldwide,
12 particularly on islands and in Australia where native species are generally naïve to the threat of
13 this introduced predator. Effectively reducing cat populations to protect wildlife is challenging
14 because cats have a cryptic nature, high reproductive rate and strong reinvasion ability.

15 *Aims* We experimentally tested the response of feral cats and their native prey to an *Eradicat*[®]
16 poison baiting program at a conservation reserve.

17 *Methods* Baits were distributed by hand along roads and tracks every 50 m (~10 baits km⁻²). We
18 used camera traps to monitor the response of cats to baiting using a repeated before-after,
19 control-impact design over six years. We also measured introduced rabbit *Oryctolagus cuniculus*
20 activity using sand pads and small mammal and reptile captures using pitfall trapping.

21 *Key results* Dynamic occupancy modelling revealed only modest effects of baiting on cats in two
22 out of six years, with occupancy in the baited area decreasing from 54% to 19% in 2014 (-35%)
23 and 89% to 63% in 2017 (-26%). Baiting effectiveness was not related to antecedent rainfall or
24 prey availability. Bait availability was reduced by non-target interference; 73% of 41 baits were
25 removed by non-target species. We found no evidence for persistent changes in small mammal
26 or reptile capture rates in the baited area relative to the unbaited area over the life of the
27 project.

28 *Conclusions* Relatively low baiting density and non-target interference with baits are likely to
29 have reduced baiting efficacy. Further testing and refinement of ground baiting is needed,
30 including trialling higher baiting densities and/or frequencies.

31 *Implications* We highlight key areas for future research that should benefit feral cat
32 management not only in Australia, but also on the many islands worldwide where cats threaten
33 native wildlife.

34
35 **Keywords:** cat baiting, dynamic occupancy model, impact evaluation, invasive predator, lethal
36 control, pest control

38 **Short summary:** Feral cats are a leading cause of biodiversity loss worldwide and effective cat
39 control remains a challenge for many land managers. We experimentally tested the response of
40 cats and their prey to a ground baiting program over six years, but found low efficacy in terms
41 of changes in cat occupancy and activity each year. Increasing baiting density and/or frequency
42 may improve the efficacy of ground baiting.

44 **Introduction**

45 Invasive predators are a major driver of global biodiversity loss, having contributed to more
46 than 50% of bird, mammal and reptile extinctions worldwide (Doherty *et al.* 2016). Their
47 impacts have been greatest on islands, where prey species are typically naïve to the threat of
48 introduced predators (Salo *et al.* 2007; Medina *et al.* 2011). Feral cats *Felis catus* are one of the
49 most damaging species because humans have spread them across the world, they are highly
50 adaptable to varying environmental conditions (Bengsen *et al.* 2016), and they prey on a range
51 of birds, mammals, reptiles, amphibians and invertebrates (Bonnaud *et al.* 2011). We use the
52 term ‘feral cats’ to refer to animals that live in the wild and have no direct dependence on
53 humans. Reducing the impacts of cats on native wildlife is a key concern of conservation
54 practitioners and scientists globally (Nogales *et al.* 2013; Shionosaki *et al.* 2015; Loss and Marra
55 2017; Doherty *et al.* 2017).

56
57 Management of feral cats has typically focussed on lethal control, including trapping, shooting,
58 and poison baiting. Each of these methods rely on removing individual predators from a system
59 to reduce or eliminate predation pressure on prey species. Cats have a high reproductive rate
60 and ability to reinvade, so lethal control must be intensive and sustained in order to effectively
61 reduce cat population densities (Leo *et al.* 2018; Lohr and Algar 2020). Demographic studies
62 indicate that, on average, more than 57% of a cat population must be removed annually to
63 reduce population densities (Hone *et al.* 2010). Achieving this in practice, though, has been
64 challenging due to the cryptic nature of cats and their aversion to entering traps and consuming
65 baits (Fisher *et al.* 2015). For instance, the huntability of cats (number of cats shot at as a
66 percentage of those sighted) on Marion Island ranged from 25–44% over four years (Bloomer
67 and Bester 1992). Low intensity trapping and removal of cats in Tasmania, Australia actually
68 caused an increase in cat activity and relative abundance at removal sites, possibly due to
69 immigration by neighbouring cats into vacated territories (Lazenby *et al.* 2014). In the Flinders
70 Ranges, South Australia, the number of cat detections on cameras did not change after 40 cats
71 were removed through trapping over two months (Stobo-Wilson *et al.* 2020). On Rota Island,
72 spotlight hunting of cats caused a modest knockdown within the first 18 months, but the
73 population stabilised over the next 11 months (Leo *et al.* 2018). These examples illustrate that

74 effective cat control can be very time and labour intensive, meaning that resource and financial
75 constraints can also hinder success.

76

77 In Australia, where cats have contributed to the extinction of more than 20 native mammal
78 species and threaten many other birds, mammals and reptiles (Woinarski *et al.* 2015; Woinarski
79 *et al.* 2019), there are two specially designed poison baits that can effectively reduce cat
80 populations, dependent on environmental conditions (e.g. Johnston *et al.* 2014; Comer *et al.*
81 2018; Lohr and Algar 2020). *Eradicat*[®] and *Curiosity*[®] are small sausage style baits comprised of
82 kangaroo meat, chicken fat, and digest and flavour enhancers (Algar *et al.* 2007; Johnston *et al.*
83 2013). *Eradicat*[®] is registered for use in parts of Western Australia and contains 4.5 mg of 1080
84 poison (sodium fluoroacetate) injected into the bait (Algar *et al.* 2007). *Curiosity*[®] is designed
85 for use in southern and eastern Australia and contains a hard capsule of para-
86 aminopropiophenone (PAPP) poison (Johnston *et al.* 2013; Johnston *et al.* 2014). *Eradicat*[®] is
87 usually deployed aerially at a rate of 50 baits km⁻², which can reduce cat populations, although
88 effectiveness varies between years (Algar *et al.* 2007; Richards and Algar 2010; Algar *et al.*
89 2011; Algar *et al.* 2013; Comer *et al.* 2018; Lohr and Algar 2020). The baits are also readily
90 consumed by dingoes *Canis dingo* and introduced red foxes *Vulpes vulpes* and thus can also
91 reduce their population densities (Richards and Algar 2010; Berry *et al.* 2014; Wysong *et al.*
92 2020b).

93

94 In addition to aerial baiting, baits can also be distributed along tracks and roads ('ground
95 baiting'), although this approach has received less attention (but see Doherty and Algar 2015;
96 Burrows *et al.* 2018; Lohr and Algar 2020; Fancourt *et al.* 2021). Ground baiting may be a more
97 accessible option for smaller landholders that do not have adequate resources for aerial
98 operations (e.g. some non-government organisations and private landholders). However,
99 because ground baiting relies on roads and tracks for bait distribution, potential baiting
100 densities are reduced, which may limit effectiveness. Placing baits along tracks may increase
101 encounter rates by animals that preferentially move along tracks (Geyle *et al.* 2020; Wysong *et*
102 *al.* 2020a), but by the same token the baits will be biased away from animals that rarely use
103 tracks (Fancourt *et al.* 2021).

104

105 In this study, we experimentally tested the impacts of annual *Eradicat*[®] ground baiting on feral
106 cat occupancy and activity over six years (2013–19, excluding 2015) at a conservation reserve
107 in Western Australia. We monitored impacts of baiting on both feral cats and their prey,
108 comparing the baited area with an unbaited control area (BACI experimental design). This
109 project began under an experimental permit before *Eradicat*[®] was registered by the national

110 regulator (Australian Pesticides and Veterinary Medicines Authority). The baiting involved an
111 annual application of baits at a density of ~ 10 per km^2 (one bait every 50 m), which is 20% of
112 that used in aerial operations. This is the maximum density that we could use and we were
113 limited to applying baits once per year under the experimental permit. Results from the first
114 two years of this project showed that baiting reduced cat activity in 2014, but not 2013
115 (Doherty and Algar 2015). We reanalyse that data here, along with additional data, to address
116 the following research questions for 2013–2019:

- 117 1. Which species consume *Eradicat*[®] baits and how often?
- 118 2. Does annual ground baiting with *Eradicat*[®] reduce cat occupancy and activity?
- 119 3. Is baiting effectiveness related to rainfall or prey availability?
- 120 4. Have capture rates of small mammals and reptiles in the baited area changed over time
121 relative to the unbaited area?

122

123 **Methods**

124 Study site and design

125 This study was conducted at Charles Darwin Reserve, a $\sim 68,000$ ha property managed for
126 conservation by Bush Heritage Australia ($-29.65, 116.97$; Fig. 1). The climate is semi-arid
127 Mediterranean and mean annual rainfall is approximately 300 mm (Wanarra weather station;
128 Bureau of Meteorology 2020). Vegetation primarily consists of *Acacia* shrublands and
129 *Eucalyptus* woodlands, with smaller areas of salt lakes, and granite outcrops and breakaways.
130 The property is surrounded by Unallocated Crown Land to the south, pastoral leases to the west
131 and north, and the Mt Gibson Wildlife Sanctuary to the east. The reserve does not have predator
132 or herbivore exclusion fences. Cats are common in the study area and dingoes less so, whereas
133 foxes are comparatively rare (Doherty 2015b).

134

135 The baited treatment area ($\sim 150 \text{ km}^2$) was located in the southern half of the reserve and the
136 unbaited experimental control area ($\sim 100 \text{ km}^2$) was located in the northern half (Fig. 1). Toxic
137 *Eradicat*[®] baits were distributed in the baited area once annually from 2013 to 2019. Baits were
138 laid by hand along alternating sides of vehicle tracks at a rate of one bait every 50 m, with
139 $\sim 1,500$ baits laid each year, resulting in a density of ~ 10 baits km^{-2} . Baits were not replaced
140 once they were removed. Baiting was conducted once in each of September 2013, May 2014,
141 June 2015, and May 2016–19. However, we do not present the 2015 data due to multiple issues
142 with camera trap operation that year that compromised data collection. As detailed below, our
143 analyses include using dynamic occupancy models and generalised linear mixed models
144 (GLMMs) to assess cat responses to baiting (question 2), general linear models to assess drivers

145 of baiting effectiveness (question 3), and GLMMs to model changes in small mammal and reptile
146 capture rates (question 4).

147

148 Camera set-up and bait uptake trials

149 Cats were monitored before and after baiting each year using motion-sensing cameras. Twenty
150 cameras each separated by ≥ 2 km were deployed in each of the baited and unbaited treatments.
151 At this spacing, the cameras are unlikely to have been spatially independent for the purposes of
152 occupancy modelling, thus the occupancy results should be interpreted as probability of site
153 use, rather than true occupancy (MacKenzie *et al.* 2018). Most of the unbaited cameras (75%)
154 were >9.50 km from the baited area, with the minimum distance being 5.50 km. We considered
155 this distance large enough for the two treatments to be considered independent for cats, given
156 that home range studies from similar Australian environments recorded mean home ranges of
157 2.48–22.10 km² (Jones and Coman 1982; Edwards *et al.* 2001; Molsher *et al.* 2005; Hilmer 2010;
158 Bengsen *et al.* 2012), which corresponds to a home range diameter of 1.80–5.30 km (if assumed
159 to be a circle). Although dingoes were recorded on our cameras, we do not present the data here
160 because the treatments were too close together to be independent for dingoes, given their much
161 larger home ranges (Harden 1985; Robley *et al.* 2010; Newsome *et al.* 2013; Allen *et al.* 2014).

162

163 The cameras used in 2013–15 were a mixture of Moultrie i60 and Scoutguard 560PV units,
164 whereas the cameras used in 2016–19 were Reconyx HC600 Hyperfire. Equal numbers of the
165 two camera models were deployed in each treatment in 2013–15 and assignment to locations
166 was randomised, in order to reduce any bias. Cameras were fixed to steel posts ~ 30 cm above
167 the ground and next to vehicle tracks. Cameras were programmed to take three photographs
168 each time the sensor was triggered, with a minimum delay of one minute between triggers. To
169 measure bait uptake (question 1), each of the 20 cameras in the baited area had a bait placed in
170 front of it during the baiting period in 2015–19. Memory cards were collected from cameras
171 after 1–3 weeks after the baiting and photos were inspected to assess whether baits were taken,
172 which species were responsible, and how long after bait placement removal took place.

173

174 Feral cat occupancy and activity (question 2)

175 Each pre- and post-baiting monitoring session lasted for 4–6 weeks (Table S1), although some
176 individual cameras stopped working prematurely due to battery failure or memory cards
177 reaching capacity. Post-baiting monitoring began 1–4 weeks after baiting, except in 2016 when
178 it began two months after baiting. In 2013–15, half of the cameras had a scent lure and the other
179 half an audio lure, which were swapped between cameras halfway through each monitoring
180 session. Scent lures were a fresh chicken wing encased in a PVC tube pegged to the ground, with

181 a fresh chicken wing provided when lures were swapped. The audio lure was a small electronic
182 device that emitted the sound of a bird tweeting (Lucky Duck, WI, USA). Only scent lures
183 (chicken wings) were used at all cameras in 2016–19.

184

185 Photos were manually inspected, and the presence of animal species was recorded, along with
186 the site number, date and time. We created dataframes relating to sampling effort and detection
187 histories using the `camtrapR` package in R (Niedballa *et al.* 2016; R Core Team 2019). The
188 sampling effort dataframes recorded when cameras stopped functioning prior to the end of
189 survey periods. Two cameras were also stolen in 2013. Detection histories represented the
190 presence or absence of a species at each camera during each successive three-day period
191 throughout a survey. For instance, a 31-day survey would have 10 three-day sampling
192 occasions, with the extra day excluded. We chose a three-day, rather than daily, sampling period
193 to improve model convergence.

194

195 We used dynamic occupancy models in the `unmarked` package in R (Fiske and Chandler 2011)
196 to assess changes in cat occupancy in response to baiting each year. Dynamic occupancy models
197 use data from multiple primary periods (pre-baiting and post-baiting here) that are comprised
198 of multiple secondary periods (three-day blocks in this case) and do not assume that the system
199 is closed between the primary periods. This approach enables the estimation of detectability,
200 initial site occupancy (first primary period), and colonisation and extinction probabilities, which
201 represent changes in site occupancy between primary periods, i.e. from before to after baiting.

202

203 We analysed each year separately and used a multi-step approach by first conducting all subsets
204 model selection on the detection component and then fitting a specific set of hypothesis-based
205 models for the occupancy and extinction components. First, we determined whether detection
206 probabilities were influenced by vegetation composition at each site using model selection. We
207 calculated the proportion of five vegetation types within a 500 m radius of each camera: young
208 shrublands (<20 years since fire), old shrublands (≥20 years since fire), woodlands, salt lakes
209 and granite outcrops. We excluded young shrublands from the modelling because it was
210 negatively correlated with old shrublands (Pearson's $r = -0.58$) and woodlands (-0.43). We
211 fitted a global detection model containing main effects for each of the four vegetation variables,
212 while holding occupancy, colonisation and extinction probabilities constant. For 2013 and 2014,
213 we also included camera model (Moultrie or Scoutguard) as a potential predictor of
214 detectability. We used the `dredge` function in the `MuMIn` package (Bartoń 2019) to fit all
215 possible model combinations and ranked the models using Akaike's Information Criterion
216 corrected for small sample size (AICc). For the subsequent modelling of occupancy and

217 extinction probabilities, we used the detection variables from the model with the highest
218 weight, which in some cases was the null model (Table S2).

219

220 To determine whether baiting had an effect on cat occupancy, we fitted and ranked a series of
221 models that included treatment (baited/unbaited) as a predictor of initial site occupancy,
222 extinction probability, or both. These models also included the detection covariates as per Table
223 S2. We compared these three models to a null model that only included the detection covariates
224 (if applicable). Because the experiment used a before-after, control-impact (BACI) design, an
225 effect of baiting would be supported if there was an effect of treatment on extinction
226 probability. Specifically, we would expect extinction probability from pre- to post-baiting to be
227 higher in the baited compared to unbaited area. We discuss the results from any models with a
228 $\Delta AICc \leq 2$. We used parametric bootstrapping with 1,000 simulations to derive pre- and post-
229 baiting occupancy estimates for each treatment, which we present as means with 95%
230 confidence intervals.

231

232 We also used generalised linear mixed models with a Gaussian distribution to assess changes in
233 cat activity in response to baiting. The activity index was calculated by dividing the number of
234 independent detections of cats on each camera by the number of nights the camera was active
235 and multiplying this by 100 (i.e. number of detections per 100 trap-nights). Independent
236 detections were photographs on the same camera that were at least 60 minutes apart. The
237 models included fixed effects of Time (pre-/post-baiting), Treatment (baited/unbaited), and
238 Time×Treatment. A significant interaction would support an impact of baiting on cat activity.
239 Models also included a random effect of Site to account for repeat sampling. We fitted a separate
240 model for each year and present parameter estimates and 95% confidence intervals. To provide
241 further context to the occupancy and activity results, we also present in the Supplementary
242 Materials plots of naïve occupancy, which represents the proportion of sites within each
243 treatment where each species was detected, disregarding varying sampling effort.

244

245 *Drivers and outcomes of baiting effectiveness (questions 3 and 4)*

246 We used data on rainfall, small mammal capture rates, and introduced rabbit *Oryctolagus*
247 *cuniculus* activity to assess if baiting effectiveness is related to the availability of potential prey
248 (question 3). We calculated total rainfall for the six months (Rain_6m) and 12 months
249 (Rain_12m) prior to each baiting event using data from a rain gauge at the reserve. We used
250 pitfall trapping data to estimate capture rates of small mammals (Mammal_CR' number of
251 individuals captured per trap-night) in the spring (September–November) prior to each baiting
252 event (n = 8–16 sites). The small mammal species captured were *Notomys mitchellii*, *Pseudomys*

253 *hermannsburgensis*, *Mus musculus* and *Sminthopsis* spp. (see Doherty *et al.* 2015 for further
254 details). As per previous studies (e.g. Coates 2008; Weston *et al.* 2009), we used sand pads to
255 monitor rabbits and calculated an index of rabbit activity for both the spring (Rab_spr) and
256 winter (Rab_win) prior to each baiting event (i.e. in the previous year). The index was calculated
257 as the proportion of days rabbits were detected on each sand pad ($n = 69$), averaged across all
258 sand pads for each season. Sand pad data were not available for winter 2013 and spring 2012
259 and 2013. To further assess the relationship between prey availability and baiting effectiveness,
260 we calculated a ratio (PP_ratio) by dividing Mammal_CR by mean pre-baiting cat activity across
261 all cameras for each year, following Christensen *et al.* (2013). Pitfall trapping and sand pad
262 monitoring sites were spread across baited and unbaited areas (Fig. S1), although we pooled all
263 data for analysis to assess broadscale inter-annual variation in prey availability, and because
264 there were few differences between treatments across years (see Results). Full details of the
265 field sampling are provided in the Supplementary Materials.

266

267 To create a measure of baiting effectiveness for each year, we subtracted the mean difference in
268 activity or occupancy between baited and unbaited sites pre-baiting, from the mean difference
269 after baiting, i.e. $(After_{Unbaited} - After_{Baited}) - (Before_{Unbaited} - Before_{Baited})$ (Christie *et*
270 *al.* 2019). For the occupancy metric, we used the estimates from the top ranked model for each
271 year. Using this approach, values of 0 indicate that the difference between treatments is equal
272 for before and after baiting. Positive values indicate a greater difference in favour of unbaited
273 sites, i.e. occupancy decreased more in baited than unbaited sites and/or unbaited sites
274 increased more than baited sites. Negative values indicate a greater difference in favour of
275 baited sites, i.e. occupancy in unbaited sites decreased more than in baited sites and/or baited
276 sites increased more than unbaited sites. As such, more positive values are indicative of a
277 greater impact of baiting.

278

279 To test the relationship between baiting effectiveness and environmental variables, we fitted
280 general linear models with either the occupancy or activity baiting effectiveness metric as the
281 response variable and either Rain_6m, Rain_12m, Mammal_CR, PP_ratio, Rab_spr, or Rab_win as
282 the predictor variable. We included only one predictor variable per model due to the small
283 sample size ($n = 4-6$ years). We present model parameter estimates, 95% confidence intervals
284 and plots of the data.

285

286 We also used generalised linear mixed models assuming a Gaussian distribution to assess if
287 capture rates of small mammals and reptiles have changed over time in baited and unbaited
288 areas (question 4). Data were derived from the pitfall trapping as described above and in the

289 Supplementary Materials. The response variable was small mammal or reptile capture rate for
290 each spring 2012–18 (excluding 2014, when sampling was not undertaken). 2012 represents a
291 baseline from before baiting began. We included fixed effects of Year, Treatment, and
292 Year×Treatment, and a random effect of Site to account for repeat sampling.

293

294 **Results**

295 Bait uptake

296 Of the 100 baits laid in front of cameras in 2015–19, 30 could not be seen on camera due to poor
297 placement or camera malfunction, and a further 29 baits disappeared without the event being
298 recorded on camera. Of the remaining 41 baits, we recorded an animal interaction for 34 baits.
299 Records of cats included one bait consumed nine hours after being laid, another bait inspected
300 but not removed seven days after being laid, and another bait that a cat walked past without
301 seeming to inspect it, three days after it was laid. Twenty baits were either eaten or taken away
302 by emus *Dromaius novaehollandiae* (49% of visible baits), seven by ravens *Corvus coronoides*
303 (17%), and one each by a fox, hopping mouse *Notomys mitchellii*, and grey currawong *Strepera*
304 *versicolor*. Emus removed baits within 0–9 days of baits being laid (mean = 4.05). In one case,
305 the bait was removed five hours after being laid. Ravens removed baits within 1–22 days (mean
306 = 6.43) and the fox, hopping mouse and currawong removed baits three days, two days and five
307 minutes after being laid, respectively. Twenty-six minutes after the fox consumed a bait, a fox
308 also inspected but did not remove a bait at a neighbouring camera. One dingo was also recorded
309 walking past a bait without seeming to inspect it. Ten baits remained in place at the end of the
310 trial period.

311

312 Occupancy and activity

313 Over the six-year study, feral cats were detected at 3–18 treatment cameras and 12–19 control
314 cameras each year (mean = 11.67 and 14.83 of 20 cameras, respectively). There was uncertainty
315 in the impacts of baiting on cat occupancy in most years, with multiple models having $\Delta AICc$
316 values ≤ 2 (Table S4, Fig. S2). There was only one year with a clear impact of baiting on cat
317 occupancy: in 2017 occupancy decreased in the baited area post-baiting (from 0.89 to 0.63 in
318 the top model), whereas it increased slightly in the unbaited area (0.89 to 1.00; Fig. 2). The
319 changes in activity and naïve occupancy also support an impact of baiting in 2017, including a
320 significant Time×Treatment interaction in the mixed model (Fig. 2 & S3, Table S5).

321

322 The best supported occupancy models for 2014 and 2019 also indicated greater declines in
323 occupancy in the baited compared to unbaited area (Fig. 2), although the null models were also
324 well supported (Table S4, Fig. S2). Changes in naïve occupancy and the activity index support an

325 impact of baiting in 2014, but not 2019 (Fig. 2 & S3). The Time×Treatment interaction for
326 activity in 2014 had confidence intervals slightly overlapping zero (-0.48–5.56; Table S5).
327 Estimated occupancy from the top model for 2014 was 0.54 in both treatments pre-baiting and
328 0.48 in the unbaited area and 0.19 in the baited area post-baiting (Fig. 2). The activity
329 confidence intervals for Time×Treatment in 2013 and 2018 also only slightly overlapped zero
330 (Table S5), but when considered together with the occupancy data, there is little support for an
331 impact of baiting.

332

333 *Drivers and outcomes of baiting effectiveness*

334 Pre-baiting occupancy, activity and naïve occupancy were similar between baited and unbaited
335 treatments each year from 2014 to 2019 (Fig. 2 & S3), indicating that there was no cumulative
336 effect of baiting on cat activity or occupancy over time. There was no relationship between
337 baiting effectiveness (based on changes in cat activity/occupancy) and rainfall, small mammal
338 capture rate, rabbit activity, or the predator-prey ration index (Table S6, Fig. S4). Small mammal
339 capture rates were similar between treatments every year, except in 2015 and 2016 when
340 average capture rates were 2.1- and 1.8-fold higher, respectively, at baited compared to
341 unbaited sites (Fig. 3, Table S7). Mean capture rates of reptiles did not vary between treatments,
342 but overall capture rates in 2013 and 2016–18 were lower than the baseline year of 2012 (Fig.
343 3, Table S7).

344

345 **Discussion**

346 We experimentally tested the impact of poison baiting on feral cats and potential benefits for
347 their native prey at a conservation reserve over six years. Annual ground baiting using
348 *Eradicat*[®] at ~ 10 baits km⁻² was mostly ineffective at reducing cat occupancy, with there being
349 only weak treatment effects in two out of the six years tested (absolute decreases in occupancy
350 of 35% in 2014 and 26% in 2017). Consistent with the limited effect of baiting on feral cats, we
351 found no evidence of benefits for their native prey. There are a number of factors that could
352 have limited the efficacy of the baiting program, including baiting density, prey availability and
353 non-target uptake of baits.

354

355 The baiting density achieved in this project was ~10 baits km⁻², which is much lower than the
356 rate of 50 baits km⁻² used in aerial operations. The lower baiting density may be insufficient for
357 the majority of cats to detect baits or to encounter a bait when they are likely to eat it. At
358 Matuwa in central Western Australia, ground baiting at ~2.80 baits km⁻² was less effective than
359 aerial baiting at 50 baits km⁻² (Lohr and Algar 2020). In the Gibson Desert, Burrows et al. (2003)
360 found that a fresh meat bait designed for cats reduced activity by 75% and 100% at densities of

361 10 and 22 baits km⁻², respectively, during low rainfall periods, whereas baiting at 11 baits km⁻²
362 during a high rainfall period reduced activity by 25%. Ground application of fresh meat baits at
363 a density of 7.3 km⁻² was ineffective at reducing cat densities in Queensland (Fancourt *et al.*
364 2021). In South Australia, Moseby & Hill (2011) tested aerial *Eradicat*[®] baiting at densities of 10
365 and 25 baits km⁻² and found that cat activity declined in response to baiting in just one of seven
366 trials. When taken together, these studies suggest that a baiting density of 10 baits km⁻² is not in
367 itself insufficient, but may be when combined with other factors, as discussed below.

368

369 The effective baiting density in our study was likely reduced by non-target consumption of
370 baits. Our uptake trials in 2013–14 (reported in Doherty and Algar 2015) showed that corvids
371 *Corvus* spp. removed the most baits (12 of 30), followed by cats (6) and varanids *Varanus* spp.
372 (2). In 2015–19, 73% of the baits where fate could be determined were removed by non-target
373 species, primarily emus, which removed 49% of visible baits. We only recorded one bait being
374 removed by a cat. Our effective sample size was modest (41 baits) because fate could not be
375 determined for a large number of baits, but if we assume that none of those baits were taken by
376 non-targets (which is unlikely), then an absolute minimum of 30% of baits were removed by
377 non-target species. Of the baits removed by non-targets, 47% were removed within three days
378 of being laid and 90% within seven days. This means that the window of bait availability to cats
379 is very narrow and when combined with the already low propensity of cats to consume baits,
380 the chances of bait uptake are very low.

381

382 Many other studies have also recorded high non-target uptake of cat baits, including 22% of
383 baits at Peron Peninsula (Algar *et al.* 2007), 14–57% at Arid Recovery (Moseby *et al.* 2011),
384 71% at Kangaroo Island (Hohnen *et al.* 2020), and 97% at Dryandra and Tutanning (Friend *et al.*
385 2020). Only one of those studies recorded emus removing baits (Algar *et al.* 2007), although it is
386 not clear what proportion of baits were interfered with. Emus commonly travel long distances
387 along tracks at our study site and elsewhere (TSD, *personal observation*), which would provide
388 them with ample opportunity to encounter baits, which were placed at 50-m intervals.
389 However, it is also worth noting that baits for the uptake trials were placed in the open to
390 facilitate camera monitoring, which may have increased their detection and hence removal by
391 non-target species. Moseby *et al.* (2011) found that bait removal by corvids was higher for baits
392 in the open compared to those under bushes. In practice, most of the 1,500 baits deployed each
393 year were placed underneath or close to shrubs, thus actual rates of non-target removal may
394 have been lower. We recommend that future work quantify rates of non-target removal for baits
395 that are placed in the open and under shrubs, and on and off tracks.

396

397 Prey availability is one of the strongest determinants of the efficacy of cat baiting programs,
398 which itself is primarily driven by rainfall in arid and semi-arid Australia (Letnic and Dickman
399 2010). Using data from three sites in Western Australia, Christensen et al. (2013) showed that
400 the efficacy of aerial cat baiting was predicted by a predator-prey ratio index. Reductions in cat
401 activity were greatest when the amount of prey available per cat was lowest (as inferred from
402 indices; Christensen *et al.* 2013). Short et al. (1997) also found that bait uptake by cats was
403 inversely related to rabbit density. In contrast, we found no relationship between baiting
404 effectiveness and a similar predator-prey index, nor with measures of rainfall, rabbit activity, or
405 small mammal capture rate. However, this does not mean that prey availability is not important
406 in our study area. It may just be that small sample sizes and low bait availability (and hence
407 effectiveness) inhibited the detection of any trends. A larger sample size covering a greater
408 range of baiting effectiveness would provide more detailed insights.

409

410 Other factors that could have limited the efficacy on ground baiting in this study include the
411 relatively small area that was baited and the fact that baits were only applied once per year. The
412 size of our baited area (~15,000 ha) was dictated by the need to fit both the treatment and
413 control areas within the one property. Over time, any effects of baiting would have been
414 reversed as cats from surrounding unbaited areas reinvaded the baited area (Algar *et al.* 2013;
415 Lazenby *et al.* 2014). However, this does not preclude the detection of short-term effects of
416 baiting (within 1–2 months) and we did indeed detect modest effects of baiting in two out of six
417 years. Increasing the frequency at which baits are laid may improve the chances that cats
418 encounter and consume baits. However, only a single annual application of baits was permitted
419 under our experimental permit, and autumn baiting is recommended in the semi-arid zone due
420 to lower prey availability at that time of year. Applying baits more frequently during this time
421 could increase the amount of time that fresh baits are available and thus the likelihood that cats
422 encounter baits when they are hungry or are using tracks (Algar *et al.* 2007).

423

424 A strength of this study was the inclusion of an unbaited reference area for comparison,
425 allowing treatment effects to be separated from seasonal effects that could otherwise bias
426 conclusions. For instance, if we did not survey an unbaited control area, we could have
427 concluded that baiting was effective in 2018 and 2019, even though decreases in activity and
428 occupancy were similar between baited and unbaited areas. We recommend that future work,
429 where feasible, include an unbaited reference area to maximise inferential strength (Christie *et*
430 *al.* 2019). This is especially important where control techniques are still being developed, or
431 widespread and consistent effectiveness has not yet been demonstrated. However, where

432 inclusion of an unbaited reference area is not feasible, GPS/VHF tracking of target animals to
433 record survival post-baiting can provide complementary information about baiting efficacy.

434

435 Controlling predators is only a means to an end and the ultimate aim of any invasive predator
436 control program should be to produce positive outcomes for the target asset, such as increased
437 species richness, abundance, or threatened species survival. Small mammals and reptiles, along
438 with rabbits, are the major components of cat diets in our study area (Doherty 2015a). We
439 found no consistent differences in capture rates of small mammals and reptiles between baited
440 and unbaited areas over the life of this project. Small mammal captures were higher in baited
441 compared to unbaited areas in 2015 and 2016, but since we were unable to measure the impact
442 of baiting on cats in 2015, it is difficult to contextualise these results. However, the overall lack
443 of differences is not surprising given that there were only modest effects of baiting on cats in
444 two out of six years (2014 and 2017). For cat control to benefit prey populations, reductions in
445 cat populations must be intense and sustained, although there is little precise information about
446 the minimum level of suppression required (Norbury *et al.* 2015). The clearest examples are
447 where cats (and foxes) have been eradicated from islands and fenced exclosures, resulting in
448 dramatic improvements in the survival of reintroduced mammals (Legge *et al.* 2018) and
449 increases in the abundance and richness of resident fauna (Moseby *et al.* 2009; Roshier *et al.*
450 2020). Although there are many studies demonstrating reductions in cat activity or density in
451 response to baiting, there is very little evidence available regarding the outcomes for prey
452 populations (but see Stewart *et al.* 2019; Comer *et al.* 2020). We recommend that control
453 programs for cats—and other pest species more generally—incorporate clear objectives and
454 monitoring programs for the species they are trying to protect (see also Reddiex and Forsyth
455 2006). Such monitoring can enable assessment of return-on-investment and adaptive
456 management to refine interventions over time, including detecting and addressing unexpected
457 negative consequences (Ruscoe *et al.* 2011; Walsh *et al.* 2012).

458

459 Feral cat management is challenging, and the effectiveness of different approaches varies
460 between locations and years. Further work is clearly needed to improve the effectiveness of
461 ground baiting of feral cats. Key areas for future research are investigation of methods to reduce
462 non-target removal of baits; investigation of methods that improve bait detection and
463 consumption by cats; and assessment of different baiting densities and frequencies.

464 Furthermore, since effective baiting may selectively target only a subset of the population (e.g.
465 poor hunters or bait naïve cats; Lohr and Algar 2020), complementing baiting with other
466 control methods such as trapping and shooting may be important to maximise the impacts of
467 control efforts and to reduce the risk of selecting for bait resistance (Allsop *et al.* 2017; Lohr and

468 Algar 2020). In addition, research on the effects of managing the environment by either
469 reducing the resources supporting cat populations (e.g. rabbit populations; McGregor *et al.*
470 2020), or managing fire and grazing to conserve habitat structure that provides refuges for
471 native species (Leahy *et al.* 2016; Legge *et al.* 2019), could help identify the most effective
472 approaches for feral cat management. Such work would benefit feral cat management not only
473 in Australia, but also on the many islands worldwide where cats threaten native wildlife.

474

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488

489 **Data availability** Data and code are available at FigShare

490 <https://doi.org/10.6084/m9.figshare.13240637.v1>

491

492 **Authors' contributions** TSD conceived and designed the research; all authors collected the
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494

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496

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729

730 **Figure captions**

731 **Figure 1.** Map of the study site and camera trap locations, with the smaller map showing
732 location within Western Australia.

733

734 **Figure 2.** Feral cat occupancy (top row) and activity (bottom row) for the baited and unbaited
735 areas each year. Symbols represent means and error bars represent 95% confidence intervals.

736 The dotted boxes around 2014 and 2017 represent years where the data support an impact of
737 baiting.

738 **Figure 3.** Capture rates (number of individuals captured per trap per night) of small mammals
739 and reptiles at baited and unbaited pitfall trapping sites. Symbols represent treatment means
740 and vertical lines are 95% confidence intervals.

741 **Supplementary materials**742 **Table S1.** Dates of monitoring periods and baiting in each year of the project.

Year	Pre-baiting	Baiting	Post-baiting
2013	9 Aug – 7 Sep	8 Sep	1 Oct – 6 Nov
2014	2 Apr – 9 May	11 May	20 May – 18 Jun
2015*	8 Apr – 8 May	5 June	19 Jun – 1 Aug
2016	30 Mar – 4 May	12 May	9 Jul – 7 Aug
2017	10 Apr – 12 May	19 May	26 May – 26 Jun
2018	3 Apr – 2 May	8 May	15 May – 13 Jun
2019	3 Apr – 2 May	8 May	15 May – 13 Jun

*As described in the methods, we do not present or analyse the 2015 data, but include the dates here for comprehensiveness.

743

744 **Table S2.** Detection covariates used in cat occupancy modelling.

Year	Covariates
2013	Shrub_old
2014	Shrub_old + Woodland
2016	Shrub_old + Salt_lake
2017	None
2018	Shrub_old + Salt_lake
2019	None

745

746 Additional details about pitfall trapping and sand pad monitoring

747 We used pitfall trapping data to estimate capture rates of small mammals and reptiles in the
748 spring (September–November) prior to each baiting event. Small mammals were sampled at 6–
749 16 pitfall trapping sites each year. Sites were split between young (8 to 20 years since last fire)
750 and old (26 to >55 year since last fire) shrublands (Table S3, Fig. S1). Each site consisted of two
751 parallel 25-cm high aluminum drift fences 60 m in length and separated by ~30 m. Six pitfall
752 traps (4 x 20-L buckets and 2 x 15-cm diameter PVC pipes) were situated at 10-m intervals
753 along the fences. Sites were sampled for 10 nights each in 2012 and 2013, and 4–5 nights in
754 2015–18 (mean = 4.3), so we truncated the earlier data to the first five nights of trapping. We
755 calculated small mammal capture rates for each year as the number of individuals captured at
756 each site divided by the number of trap-nights (number of nights × number of traps).

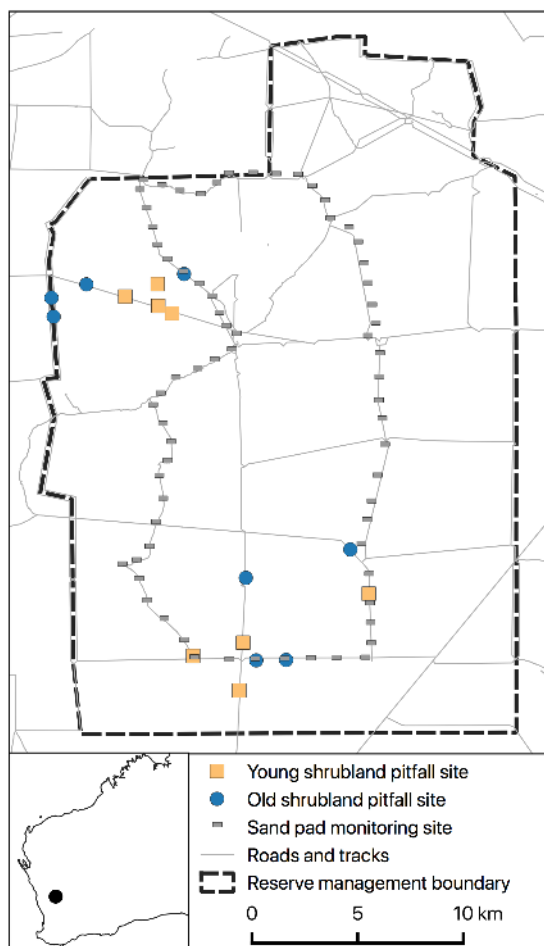
757

758 **Table S3.** Number of pitfall trapping sites in young (8 to 20 years since last fire) and old (26
759 to >55 year since last fire) shrublands sampled each year.

Year	Young shrublands	Old shrublands	Total sites
2012	8	8	16
2013	8	8	16
2015	4	4	8
2016	8	7	15
2017	7	6	13
2018	8	7	15

760

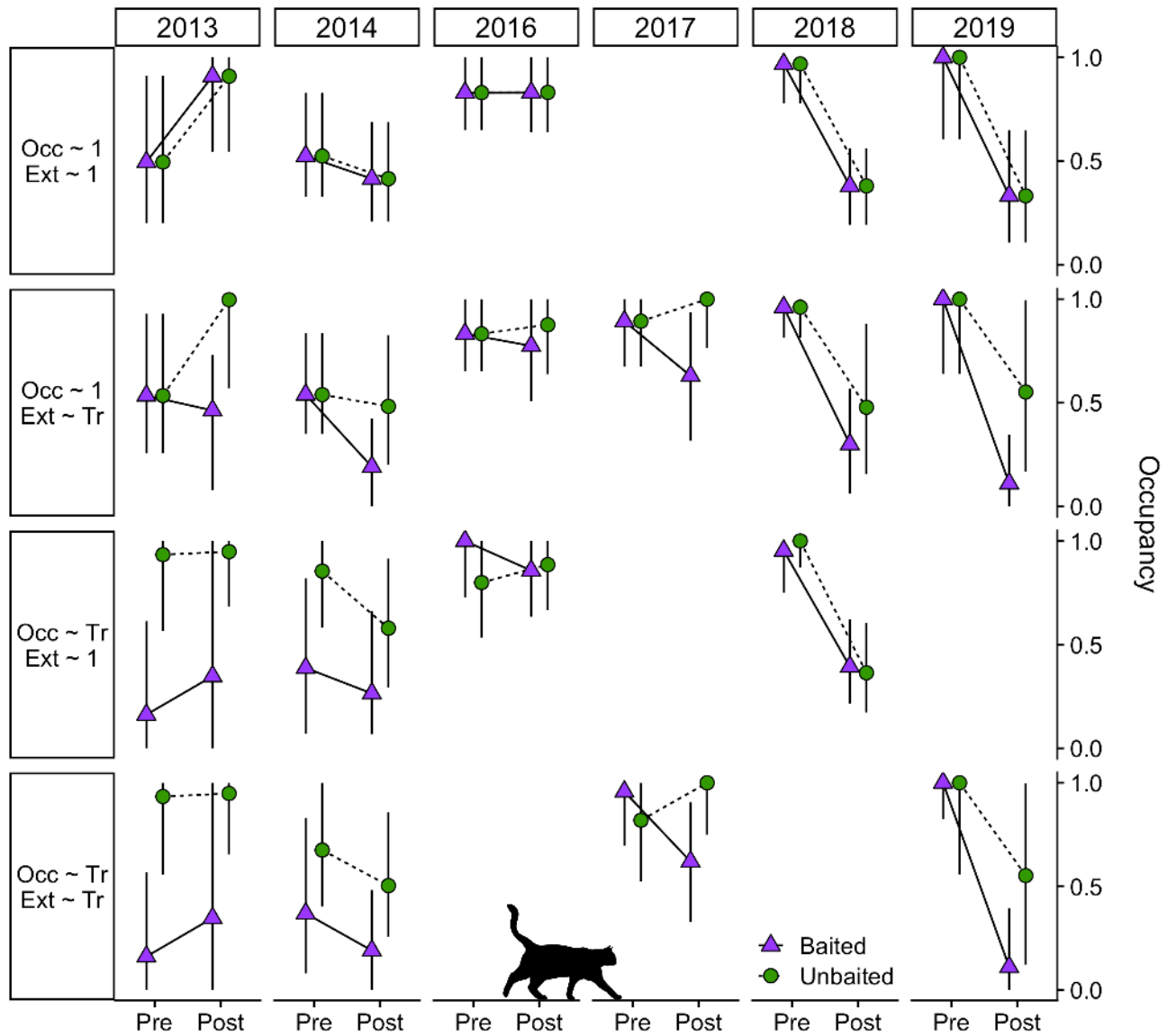
761 We used sand pad monitoring data to calculate an index of rabbit activity for both the spring
762 and winter seasons prior to each baiting event (i.e. in the previous year). There was a circuit of
763 69 sand pads separated by 1–2 km each (Fig. S1), which were monitored for 3–6 days in each
764 season and the presence/absence of rabbits and other animals was recorded each morning. The
765 index was calculated as the proportion of days rabbits were detected on each sand pad,
766 averaged across all sand pads. Sand pad data were not available for winter 2013 and spring
767 2012 and 2013.



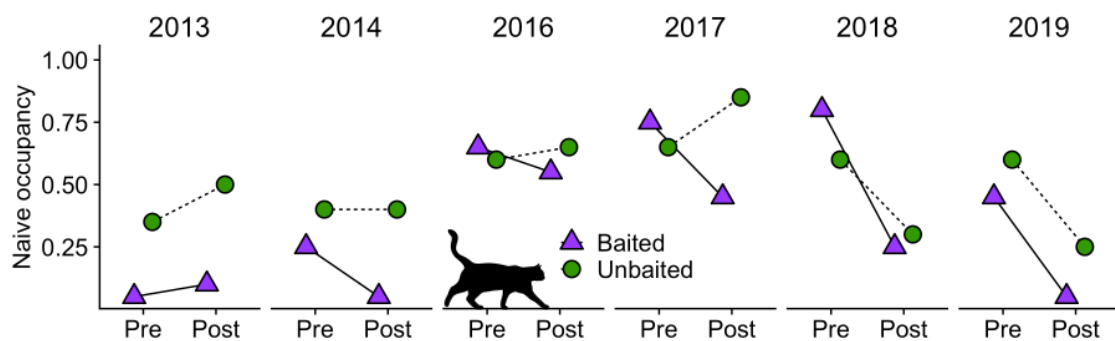
768 **Figure S1.** Map of the study site, pitfall trapping sites and sand pads, with the smaller map
769 showing location within Western Australia.

770 **Table S4.** Dynamic occupancy modelling results for feral cats. Only models with a $\Delta\text{AICc} \leq 2$ are
 771 shown. ψ , initial occupancy; ε , extinction probability.

Year	Model	ΔAICc	Weight
2013	$\psi \sim \text{Treatment}, \varepsilon \sim 1$	0	0.44
	$\psi \sim 1, \varepsilon \sim \text{Treatment}$	1.52	0.21
	$\psi \sim 1, \varepsilon \sim 1$	1.69	0.19
	$\psi \sim \text{Treatment}, \varepsilon \sim \text{Treatment}$	2.00	0.16
2014	$\psi \sim 1, \varepsilon \sim \text{Treatment}$	0	0.29
	$\psi \sim 1, \varepsilon \sim 1$	0.10	0.28
	$\psi \sim \text{Treatment}, \varepsilon \sim 1$	0.57	0.22
	$\psi \sim \text{Treatment}, \varepsilon \sim \text{Treatment}$	0.72	0.20
2016	$\psi \sim 1, \varepsilon \sim 1$	0	0.48
	$\psi \sim \text{Treatment}, \varepsilon \sim 1$	1.55	0.22
	$\psi \sim 1, \varepsilon \sim \text{Treatment}$	1.71	0.20
2017	$\psi \sim 1, \varepsilon \sim \text{Treatment}$	0	1.56
	$\psi \sim \text{Treatment}, \varepsilon \sim \text{Treatment}$	1.40	0.28
2018	$\psi \sim 1, \varepsilon \sim 1$	0	0.45
	$\psi \sim 1, \varepsilon \sim \text{Treatment}$	1.16	0.25
	$\psi \sim \text{Treatment}, \varepsilon \sim 1$	1.86	0.18
2019	$\psi \sim 1, \varepsilon \sim \text{Treatment}$	0	0.47
	$\psi \sim 1, \varepsilon \sim 1$	1.40	0.23
	$\psi \sim \text{Treatment}, \varepsilon \sim \text{Treatment}$	2.00	0.17



772 **Figure S2.** Estimated cat occupancy for all well supported models ($\Delta AICc \leq 2$) in each year. Occ,
 773 initial occupancy; Ext, extinction probability; Tr, treatment (baited/unbaited). Symbols
 774 represent means and bars represent 95% confidence intervals.
 775



776 **Figure S3.** Naïve occupancy of cats (proportion of sites with cats present) pre- and post-baiting
 777 each year.

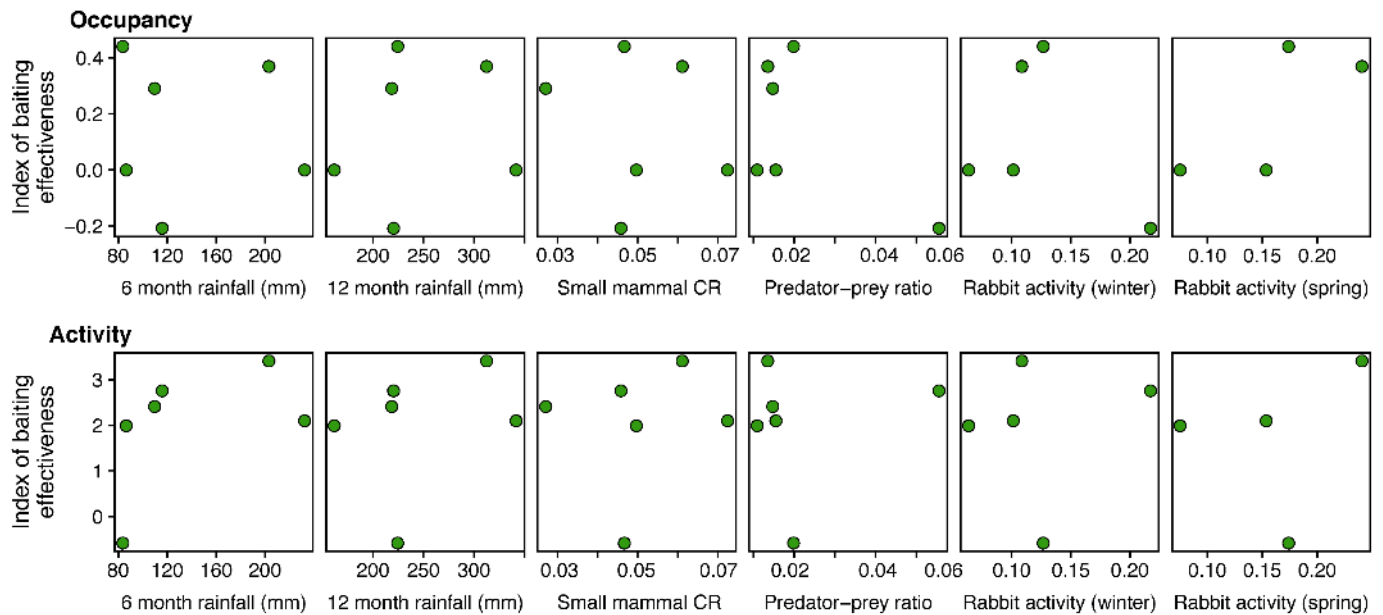
778 **Table S5.** Mixed modelling results for changes in cat activity in response to poison baiting.
 779 Values in cells are model parameter estimates and 95% confidence intervals are provided in
 780 parentheses. Time represents before or after baiting, Treatment represents baited or unbaited,
 781 and Interaction represents the interaction those two fixed effects. Bold text indicates variables
 782 where the 95% confidence intervals do not overlap zero.

Year	Intercept	Time	Treatment	Time×Treatment
2013	0.19 (-1.36 - 1.74)	0.10 (-2.06 - 2.26)	1.21 (-0.92 - 3.35)	2.76 (-0.24 - 5.76)
2014	1.17 (-0.86 - 3.21)	-1.00 (-3.14 - 1.13)	1.28 (-1.57 - 4.13)	2.54 (-0.48 - 5.56)
2016	5.04 (2.97 - 7.11)	-2.10 (-4.34 - 0.14)	-0.80 (-3.70 - 2.09)	2.77 (-0.61 - 6.16)
2017	4.79 (3.06 - 6.52)	-2.84 (-5.18 - -0.50)	-0.55 (-2.96 - 1.87)	3.44 (0.18 - 6.70)
2018	5.50 (4.08 - 6.92)	-4.17 (-6.18 - -2.16)	-1.99 (-4.03 - 0.05)	1.99 (-0.87 - 4.85)
2019	1.53 (0.49 - 2.57)	-1.37 (-2.70 - -0.03)	1.64 (0.17 - 3.11)	-0.56 (-2.45 - 1.32)

783
 784 **Table S6.** General linear modelling results for the effect of environmental variables on baiting
 785 effectiveness. Values in cells are model parameter estimates and 95% confidence intervals are
 786 provided in parentheses. Rain_6m, total rainfall for the six months prior to baiting; Rain_12,
 787 total rainfall for the 12 months prior to baiting; Mammal_CR, capture rate of small mammals for
 788 spring prior to baiting; PP_ratio, ratio of prey availability to predator activity (see Methods);
 789 Rab_win, rabbit activity index for winter prior to baiting; Rab_spr, rabbit activity index for
 790 spring prior to baiting.

Predictor	Occupancy	Activity
Rain_6m (n = 6)	-0.0002 (-0.004 - 0.004)	0.01 (-0.01 - 0.03)
Rain_12m (n = 6)	0.001 (-0.003 - 0.004)	0.01 (-0.01 - 0.02)
Mammal_CR (n = 6)	-3.04 (-19.07 - 12.99)	11.77 (-75.10 - 98.64)
PP_ratio (n = 6)	-9.08 (-20.98 - 2.81)	10.93 (-68.19 - 90.05)
Rab_win (n = 5)	-1.80 (-6.84 - 3.23)	3.06 (-26.77 - 32.89)
Rab_spr (n = 4)	2.54 (-0.74 - 5.81)	5.68 (-27.44 - 38.81)

791



792 **Figure S4.** Relationships between baiting effectiveness and environmental variables. The top
 793 row relates to occupancy and the bottom row relates to activity. Values of 0 for the baiting
 794 effectiveness index indicate that the difference between treatments is equal for before and after
 795 baiting. Positive values indicate a greater difference in favour of control sites, i.e. impact sites
 796 decreased more than control sites and/or control sites increased more than impact sites.
 797 Negative values indicate a greater difference in favour of impact sites, i.e. control sites
 798 decreased more than impact sites and/or impact sites increased more than control sites.

799 **Table S7.** Generalised linear mixed modelling results relating to differences in capture rates
 800 (CR) of small mammals and reptiles in response to Year, Treatment and the interaction. Values
 801 in cells are model parameter estimates and 95% confidence intervals are provided in
 802 parentheses. Bold text indicates predictor variables where the 95% confidence intervals do not
 803 overlap zero. The reference levels used for the intercept were 2012 and Baited.

Predictor	Level	Mammal_CR	Reptile_CR
Intercept	2012, Baited	0.03 (0.00 – 0.06)	0.24 (0.19 – 0.29)
Year	2013	-0.01 (-0.05 – 0.04)	-0.10 (-0.17 – -0.04)
	2015	0.08 (0.02 – 0.13)	-0.07 (-0.16 – 0.02)
	2016	0.05 (0.008 – 0.09)	-0.19 (-0.19 – -0.05)
	2017	0.02 (-0.02 – 0.06)	-0.20 (-0.27 – -0.13)
	2018	0.02 (-0.02 – 0.06)	-0.17 (-0.24 – -0.10)
Treatment	Unbaited	0.03 (-0.01 – 0.07)	-0.05 (-0.12 – 0.02)
Interaction	2013×Unbaited	-0.02 (-0.08 – 0.04)	0.05 (-0.05 – 0.14)
	2015×Unbaited	-0.09 (-0.16 – -0.02)	0.10 (-0.02 – 0.22)
	2016×Unbaited	-0.06 (-0.12 – -0.01)	0.01 (-0.08 – 0.10)
	2017×Unbaited	-0.03 (-0.09 – 0.03)	0.04 (-0.06 – 0.13)
	2018×Unbaited	-0.04 (-0.10 – 0.02)	0.08 (-0.01 – 0.17)

804