

1 **Estimating the benefit of quarantine: eradicating invasive cane toads from islands**

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## 21 **Summary**

- 22 **1.** Islands are increasingly used to protect endangered populations from the negative impacts  
23 of invasive species. Quarantine efforts are particularly likely to be undervalued in  
24 circumstances where a failure incurs non-economic costs. One approach to ascribe value  
25 to such efforts is by modeling the expense of restoring a system to its former state.
- 26 **2.** Using field-based removal experiments on two very different islands off northern  
27 Australia separated by > 400 km, we estimate cane toad densities, detection probabilities,  
28 and the resulting effort needed to eradicate toads from an island, and use these estimates  
29 to examine the financial benefit of cane toad quarantine across offshore islands  
30 prioritized for conversation management by the Australian federal government.
- 31 **3.** We calculate density as animals per km of freshwater shoreline, and find striking  
32 concordance of density across our two island study sites: a mean density of 353 [286,  
33 446] individual toads per kilometer on one island, and a density of 366 [319, 343] on the  
34 second. Detection probability differed between the two islands.
- 35 **4.** Using a removal model and the financial costs incurred during toad removal, we estimate  
36 that eradicating cane toads would, on average, cost between \$9444 (based on Horan  
37 Island; high detectability) and \$18093 AUD (Indian Island; low detectability) per km of  
38 available freshwater shoreline.

39       **5.** Across islands that have been prioritized for conservation benefit within the toads’  
40           predicted range, we provide an estimate of the value of toad quarantine on each island,  
41           and estimate the net value of quarantine efforts to be between \$27.25 – \$52.20 Million  
42           AUD. We explore a proposed mainland cane toad containment strategy – to prevent the  
43           spread of cane toads into the Pilbara Bioregion, and estimate its potential value to be  
44           between \$33.79 – \$64.74 M AUD.

45       **6.** *Synthesis and applications.* We present a modelling framework that can be used to  
46           estimate the value of preventative management, via estimating the length and cost of an  
47           eradication program. Our analyses suggest that there is substantial economic value in  
48           cane toad quarantine efforts across Australian offshore islands and a proposed mainland  
49           toad containment strategy.

50

51       **Key-words:** Cane Toad, density, detection probability, eradication, islands, quarantine

52

## 53 **Introduction**

54 It is a truth universally acknowledged that an ounce of prevention is worth a pound of cure. In  
55 invasive species management, this can be achieved by preventing human-mediated dispersal of  
56 non-indigenous species (Chen *et al.* 2018), by conducting routine surveillance programs aimed at  
57 early detection (Holden *et al.* 2015), and via translocation of endangered taxa beyond the current  
58 or predicted distributions of invaders (Woinarski *et al.* 2014; Legge *et al.* 2018; Moseby 2018)  
59 (National Species Management Plan, 2008). Despite such truisms, conservation managers rarely  
60 ascribe value to preventative management. Whilst preventative measures are increasingly being  
61 adopted to save imperiled taxa (Burns *et al.* 2012; Commonwealth of Australia 2015), without  
62 valuation, we risk falling prey to cognitive biases (e.g., immediacy bias), and so routinely  
63 commit substantially more money and effort to tactical, “cure” type approaches, than to strategic  
64 “prevention”. Quarantine against invasive species is a case in point; vastly more resources are  
65 spent controlling the spread and impact of invaders than are spent on preventing their arrival and  
66 establishment (Hoffman & Broadhurst 2016).

67 Quarantine is particularly likely to be undervalued in circumstances in which a failure incurs  
68 non-economic costs (e.g., biodiversity loss) (Leung *et al.* 2002) or when costs or damages persist  
69 over long-time scales (Epanchin-Neill *et al.* 2015). One way to place a value on such quarantine  
70 efforts is to calculate the cost of restoring the system to its former state (Kimball *et al.* 2014;  
71 Rohr *et al.* 2016). In the case of an invasive species with primarily non-economic impacts, we  
72 can calculate the ongoing benefit of quarantine as the expense of restoring the system to this  
73 former state, i.e., a subsequent eradication program. Such a valuation is a lower bound on the  
74 benefit of quarantine for a number of reasons. First, the same quarantine effort typically protects  
75 against many potential invasive species. In addition, any impact that an invasive species has

76 before it is eradicated (e.g., local extinction or shifts in population structure of a native species,  
77 altered landscape vegetation profiles formation) must be added to the cost of restoration  
78 (Hoffmann & Broadhurst 2016, Jardine & Sanchirico 2018). Thus, the cost of eradicating a  
79 single invader is a very conservative estimate of the true value of quarantine efforts.

80 Islands are important resources for conservation quarantine because they offer a natural barrier to  
81 the spread of invasive species. Conservation biologists routinely exploit this property of islands,  
82 not only to protect species that naturally occur on islands, but also to provide refuge for species  
83 under threat on the mainland (Thomas 2011; Tershy *et al.* 2015; Legge *et al.* 2018). In Australia  
84 alone, a minimum of 47 conservation translocations to islands have been carried out to date  
85 (Department of the Environment, Water, Heritage and the Arts, 2009). In these circumstances –  
86 where the conservation value of an island has been artificially bolstered – the subsequent arrival  
87 of invasive species can have a larger impact than they otherwise would. Typically, island  
88 quarantine is used by conservation managers to protect native species from invasive predators  
89 (e.g., foxes, cats, weasels, rats). In Australia, however, islands are also used to mitigate the  
90 impact of cane toads (*Rhinella marina*) on native predators (Moro *et al.* 2018; Ringma *et al.*  
91 2018). Cane toads were introduced to northeastern Australia in the 1930s and, in northern  
92 Australia, continue to spread westerly at a rate of ~50 km per year (Phillips *et al.* 2010). This  
93 invasion has had major impacts on populations of native predators, many of which have no  
94 resistance to the toad's toxin (Nelson *et al.* 2010; Greenlees *et al.* 2010; Llewelyn *et al.* 2014). In  
95 response to declines of multiple predator species (e.g., dasyurids, monitors, snakes) the  
96 Australian government implemented the Cane Toad Threat Abatement Plan (2011), which aimed  
97 to identify, and where possible reduce, the impact of cane toads on native species  
98 (Shanmuganathan *et al.* 2010). A lack of viable methods for broad-scale control, however, has

99 since led the Australian government to place an increased emphasis on containment (on the  
100 mainland) and on quarantine (on offshore islands) to mitigate the biodiversity impacts of cane  
101 toads.

102 While quarantine is currently the best available strategy, it is not a panacea: cane toads have  
103 already established on at least 48 islands across northern Australia (McKinney *et al.* 2018 unpub  
104 data), with potential for further natural and anthropogenic introductions. Thus, execution of the  
105 strategy outlined in the Cane Toad Threat Abatement Plan requires ongoing quarantine,  
106 eradication and containment efforts. Here we estimate the lower bound of the monetary value of  
107 these ongoing efforts, by quantifying the cost of eradicating cane toads from two islands in  
108 northern Australia. We approach this problem by estimating the density and detection probability  
109 of toads on each island, and use these estimates to calculate the amount of time and money it  
110 would take to remove enough toads to ensure eradication.

## 111 **Materials and methods**

### 112 **Study Area**

113 This study was carried out on two islands in northern Australia: Horan Island on Lake Argyle,  
114 Western Australia (HI) and Indian Island in the Northern Territory (II). Lake Argyle is Western  
115 Australia's largest man-made reservoir covering > 880 km<sup>2</sup> and is located within the East  
116 Kimberly region. The study site is composed of exposed spinifex-covered hilltops and sparse  
117 savanna woodland. Freshwater is available year-round, with the lake contracting from May–  
118 November. Toads are thought to have colonized islands on the lake in the wet seasons of  
119 2009/2010 (Somaweera & Shine 2012). Indian Island is an offshore island, 40 km west of  
120 Darwin. It supports predominantly savanna woodland and monsoonal vine thicket, with a large

121 ephemeral freshwater swamp located on the northern tip of the island. Depending of the  
122 magnitude of the wet season, standing water can be present in this swamp year-round or dry up  
123 by late September. Toads are thought to have colonized Indian Island via rafting events around  
124 2008. Access to Indian Island was granted by Kenbi Traditional Owners (Northern Land Council  
125 permit 82368).

## 126 **Field sampling**

127 Cane toad surveys occurred over six nights, on each island, denoted,  $t = \{0, 1, \dots, 5\}$ , during  
128 November 2017 (HI) and October 2018 (II). Surveys commenced at sundown each evening and  
129 lasted three hours, with ambient temperatures ranging from 24 – 35°C. As Horan Island occurs  
130 on a freshwater lake, the entire island was circumambulated each night by two people using  
131 headtorches; one individual focused on the higher part of the shoreline, the other on the lower  
132 shoreline. Indian Island is an oceanic island, with only a single freshwater swamp present in the  
133 dry season. This swamp was navigated each night by two people using head torches. On both  
134 islands, every toad encountered was collected and humanely killed on site in accordance with  
135 The University of Melbourne animal ethics protocol (1714277.1) and State laws regarding  
136 handling of non-native species. Each night, we recorded the number of individuals collected,  $c_t$ .  
137 Surveys were conducted immediately prior to the breeding season so that only post-metamorphic  
138 age classes were encountered.

## 139 **Statistical analysis**

140 We do not encounter every individual on a given night, and so incorporate imperfect detection.  
141 For each island, we aim to estimate three parameters:  $N_0$ , the true number of toads on the island  
142 at the commencement of surveys;  $p$ , the mean per-individual detection probability; and  $\alpha$ , the

143 length of time (in days) required to eradicate toads from our treatment areas. The number of  
144 individuals collected each night,  $c_t$ , can be considered a draw from a binomial distribution with:

$$145 \quad c_t \sim \text{Binom}(N_t, p).$$

146 Where  $N_0$ , the pre-sampling population size, is a latent variable with a mean and variance equal  
147 to  $\lambda$ , such that:

$$148 \quad N_0 \sim \text{Pois}(\lambda).$$

149 For  $t > 0$ :

$$150 \quad N_t = N_0 - \sum_0^{t-1} c_t.$$

151 The length of time required to remove a population,  $\alpha$ , from a treatment area is described via the  
152 relationship:

$$153 \quad \alpha = \left( \frac{\ln(r_{crit})}{\ln(1-p)} \right),$$

154 where,  $r_{crit}$ , the critical removal threshold, is equal to  $\frac{1}{N_0}$  (the inverse of the pre-sampling  
155 population size).

156 Models were fit with Markov chain Monte Carlo (MCMC) in JAGS v.4.6.0, run through R  
157 v3.4.1 via the package rjags v4.6.0 (Plummer & Martyn 2013). Three model chains were run for  
158 30,000 iterations, with the first 10,000 iterations discarded as a burn-in, which was sufficient for  
159 the MCMC chains to converge. Convergence was checked using the Gelman-Rubin diagnostic  
160 (Gelman & Rubin 1992); all chains produced potential scale reduction factors  $< 1.1$ , indicating  
161 convergence of chains. The remaining samples were thinned by a factor of 2, resulting in 10,000



162 samples per chain for post-processing. Minimally informative prior distributions for  $p$  and  $\lambda$   
163 were specified as uniform between 0 - 1 and 0 - 10,000 respectively.

164 We denote a successful eradication to have occurred when only a single toad remains (i.e., no  
165 further breeding pairs remain). As we assume that removal efforts take place on consecutive  
166 nights until completion, we disregard breeding and immigration.

### 167 **Cost analysis**

168 We estimate the cost of eradicating toads on our study islands based on consumable, personnel,  
169 and travel costs incurred during toad collection (see Appendix S1 in Supporting Information).  
170 Relative to most islands across northern Australia, both Horan and Indian Islands are readily  
171 accessible, thus our travel costs are modest. We assume that eradication is conducted by a fully-  
172 equipped organization; thus we do not include vehicle/boat purchase or hire (i.e., set-up costs),  
173 nor do we consider organizational in-kind associated with utilizing existing capital. Removal  
174 efforts are carried out on subsequent nights until eradication is reached; therefore, the cost  
175 associated with travel to and from our site is incurred only once. Travel costs include a \$85/hour  
176 consultant rate plus the additional costs of fuel, insurance, and vehicle maintenance (an extra  
177 \$36/hour). Thus, total travel costs are \$111/hour.

### 178 **Cost Scenarios**

179 We use our estimates of toad removal on Horan and Indian Islands (with their attendant detection  
180 probabilities) to highlight the potential benefit of quarantine efforts on a subset of high priority  
181 islands (Table 1). Our chosen islands are drawn from a list of 100 oceanic islands that the  
182 Australian Commonwealth has prioritized for conservation, due to their biodiversity value and  
183 presence of species listed under the Environment Protection and Biodiversity Conservation Act

184 (Department of the Environment and Energy [DEE], 1999). We refine this list to include only  
185 islands that are  $\geq 2$  km from the Australian mainland and occur within the potential distribution  
186 of cane toads in Australia (Kearney *et al.* 2008). For each island in our dataset, we map the  
187 length of permanent freshwater shoreline available, using either satellite maps,  
188 government/landholder records, or a combination of both – resulting in a net kilometer length of  
189 shoreline for each island in our dataset. All islands were cross-checked for the presence of  
190 cane toads via the ‘Feral Animals on Offshore Islands’ database (DEE, 2016) in addition to the  
191 presence of human settlement. In cases where islands had no permanent freshwater but did have  
192 human settlement (or known livestock presence), a one-kilometer circumference was assumed  
193 around dwellings and visible watering points.

194 In addition to the islands derived from this report, we explore the value of a potential cane toad  
195 containment strategy outlined in a revised version of the Cane Toad Threat Abatement Plan  
196 (Tingley *et al.* 2013).

197 This strategy aims to develop a ‘waterless barrier’ on the Australian mainland by excluding cane  
198 toads from artificial water bodies on cattle stations between Broome and Port Hedland in  
199 Western Australia. Using a dataset containing the presence of bore holes, cattle watering points,  
200 dams and permanent freshwater bodies in the Pilbara bioregion (see Southwell *et al.* 2017) we  
201 estimate the economic benefit of the proposed barrier. A one-kilometer circumference was  
202 applied to all waterpoints, dams and pools, in addition to a per-kilometer of shoreline rate along  
203 permanent watercourses within the region. If implemented successfully, this strategy could keep  
204 toads out of the Pilbara (and subsequent regions) – an effective quarantine of 268,00km<sup>2</sup> of the  
205 Australian mainland (see Florance *et al.* 2011; Tingley *et al.* 2013; Southwell *et al.* 2017 for  
206 further information).

## 207 **Results**

208 The number of cane toads removed from both Horan and Indian Island,  $c_t$ , declined over time  
209 (Figure 1). Across the duration of our surveys, we captured and removed a total of 1550 cane  
210 toads (1251 on HI, 299 on II). The estimated probability of detecting an individual toad on a  
211 given night differed between our two study sites (Horan Island: mean  $p$  [95% credible interval] =  
212 0.1 [0.07, 0.13]; Indian Island: 0.27 [0.22, 0.33]) (Figure 2). Given the site-specific detection  
213 probability, the estimated number of toads present at the initiation of our surveys ( $N_0$ ) was much  
214 higher on Horan Island (2681 [2171, 3393]) than on Indian Island (353 [308, 408]) (Figure 3).  
215 Horan Island – situated in a freshwater lake – has a circumference of 7.63 km, which translates  
216 to a cane toad density of 353 [286, 446] individuals per kilometer of freshwater shoreline. The  
217 freshwater source on Indian Island has a circumference of 1.04 km, translating to a density of  
218 366 [319, 343] individuals per kilometer of shoreline (Figure 4). We could also express toad  
219 density as animals per km<sup>2</sup> of island, in which case we calculate a density of individuals of 56 on  
220 II and 2852 on HI.

221 Given the posterior estimates of  $p$  and  $\lambda$ , we examine the total survey effort (in days) required to  
222 eradicate toads on both Horan and Indian Island. Inputting the distribution of  $N_0$ , leaving a single  
223 individual is equivalent to leaving the proportion,  $r_{crit} = \frac{1}{N_0}$  of the original individuals. The time  
224 to reach this point is given by  $\ln(r_{crit})/\ln(1 - p) = 75$  days [49, 110] on HI, and 19 days [10,  
225 29] on II (Figure 5).

226

227

## 228 **Cost Sensitivity**

229 Multiplying the estimate of the number of days required to achieve eradication by our removal  
230 costs suggests that \$62 407 [\$40 583, \$91 104] would be required to eradicate toads from HI.  
231 This equates to \$9 748 [\$6 339, \$14 231] per kilometer of freshwater shoreline or \$80 009 [\$52  
232 030, \$116 801] per km<sup>2</sup> of land. In contrast, the cost to eradicate toads from II is estimated to be  
233 \$18 394 [\$9 872, \$28 636], equating to \$17 737 [\$9 523, \$27 615] per kilometer of freshwater  
234 shoreline or \$2 929 [\$1 572, \$4 560] per km<sup>2</sup> of land (Figure 6).

## 235 **Benefit of quarantine on Prioritized Australian Islands**

236 Using our estimates of eradication costs per-kilometer of freshwater shoreline, we examine the  
237 economic benefit of cane toad quarantine on all toad-free islands (by jurisdiction), as well as the  
238 cost to restore all toad-inhabited islands to a toad-free state (Figure 7). The current economic  
239 benefit of quarantine on all prioritized toad-free islands is estimated to be between \$17.5 M (HI  
240 estimates) and \$33.3 M (based on II estimates). We estimate it would cost, on average, between  
241 \$2.8 M (HI) and \$5.2 M (II) to remove toads from all prioritized islands currently occupied by  
242 toads. Finally, we estimate the economic benefit of the ‘waterless barrier’ protecting the Pilbara  
243 to be between \$34.3 M (HI) and \$63.6 M (II).

## 244 **Discussion**

245 As the number of invasive species requiring management increases, practitioners must identify  
246 efficient strategies for allocating resources to various management activities. Although  
247 conventional wisdom places emphasis on prevention measures, the practice of valuing such  
248 actions in the face of non-economic costs can be challenging. Placing a monetary value on a

249 conservation benefit will most often require some value judgement as to the monetary worth of  
250 biodiversity. Using estimates of a species detectability, population density, and subsequent  
251 eradication costs we aim to prevent such value judgement when investigating the benefit of  
252 quarantine measures in combatting the impact of the invasive cane toad across Australia's  
253 prioritized offshore islands.

254

255 Despite substantial community and research effort into cane toad removal via trapping and hand  
256 capture, there are only a handful of published detection estimates for the species (Griffiths &  
257 McKay 2007). Our detection estimate is, of course, specific to the details of our survey.  
258 Nonetheless, it is surprisingly low for our large-shoreline site (Horan Island). Here, the length of  
259 shoreline meant we only passed each location once per night, and individual toads in this closed  
260 system had, on average, a 0.1 [0.07 – 0.13] chance of being seen on any given night. This  
261 contrasts with our small-shoreline site (Indian Island) where we were able to make multiple  
262 passes of the same point each night. Here, individual toads had a 0.27 [0.22 – 0.33] chance of  
263 being detected on a given survey night. Whilst individual toads are relatively easy to see when  
264 they are active, our results suggest that this might give a misleading impression of one-pass  
265 detectability.

266 We compared two density metrics: a linear density (per km) and an areal density (per km<sup>2</sup>). Our  
267 areal density estimate (2 852 individuals/ km<sup>2</sup>) is similar to estimates derived from previous  
268 studies of invasive cane toads in the Solomon Islands archipelago (1 035/km<sup>2</sup>; Pikacha *et al.*  
269 2015), the islands of Papua New Guinea (3 000/km<sup>2</sup>; Zugg *et al.* 1975; Freeland *et al.* 1986), and  
270 density estimates of an analogous invasive toad on Madagascar (3 240/km<sup>2</sup>; Reardon *et al.*  
271 2018). A single study conducted on the Australian mainland reported densities as high as 256

272 300 individuals per km<sup>2</sup> (Cohen & Alford 1993), but this estimate was predominantly of the  
273 metamorph life stage, which occurs at very high densities prior to dispersal. Metamorphs are  
274 strongly constrained to the edges of water bodies (Child *et al.* 2008), and typically suffer high  
275 mortality from predation and desiccation before reaching maturity (Ward-Fear *et al.* 2010).  
276 While an areal density would make sense in a habitat where animals are constrained by some  
277 factor that scales with area (e.g., primary productivity), it is clear that toads in northern Australia  
278 are often constrained by access to water in the dry season, and thus length of shoreline is more  
279 appropriate. Length of shoreline not only defines access to water, but also the density of  
280 infectious parasites (such as *Rhabdias pseudosphaerocephala*) that use the moist conditions and  
281 high toad densities along shorelines as opportunities for transmission ((Kelehear *et al.* 2011;  
282 2013). It is also likely that the survival rate of emergent metamorphs is dependent on length of  
283 shoreline, because this will set the density of conspecifics and so moderate the rate at which  
284 these conspecifics cannibalize each other (Pizzatto & Shine 2008). In comparing the areal and  
285 linear densities between our sites we find a large difference between sites in the areal metric, but  
286 a strikingly similar density value across sites in the linear metric. Our results suggest that across  
287 these two different systems, toads achieve a density of around 354 adults per kilometer of  
288 shoreline.

289 Because toads in dry conditions require regular re-hydration (Seebacher & Alford 2002; Tingley  
290 & Shine 2011) it is a logical step to conduct removal efforts when toads are restricted to a subset  
291 of semi-permanent hydration points during drier sections of the year (Letnic *et al.* 2015). Given  
292 the linear density metric is so concordant across sites, this is the best metric for calculating  
293 eradication costs. Certainly, if we use the areal metric we find a wide gulf in the possible  
294 eradication values in contrast to our shoreline metric (Figure 6). Costs based on the linear metric

295 had a greater correspondence between the two sites (\$9 748 and \$ 17 737 AUD per km of  
296 shoreline). Encouragingly our cost estimates appear similar to estimates derived from a  
297 successful eradication program associated with removing the American bullfrog from two  
298 locations in Canada (\$8 200 – \$23 000 CAN per kilometer of freshwater shoreline).

299 To our knowledge, there is only one instance in which the cost to eradicate cane toads from an  
300 island has been documented (Wingate 2011). Carried out on Nonsuch Island in Bermuda, this  
301 removal occurred over six years and included countless volunteer hours and an investment of  
302 \$10 000 USD (~\$14 330 AUD) to remove toads from an area of 0.6 km<sup>2</sup>. In addition, two  
303 successful eradications from extralimital mainland sites have been documented, occurring  
304 beyond the southern border of the cane toads' current range in Australia (White 2010; Greenlees  
305 *et al.* 2018). This handful of successful removals of the cane toad, mirrors a broad trend in the  
306 eradication of invasive amphibian populations globally (Adams & Pearl 2007; Kraus 2009;  
307 Beachy *et al.* 2011; Orchard 2011). As such, there is scant information available to guide policy  
308 makers and management agencies when evaluating the feasibility of implementing amphibian  
309 quarantine and eradication measures.

310 If we are to shift away from tactical, post-invasion approaches, to a preventative strategic  
311 approach, management practitioners require an estimate of the economic value that quarantine  
312 holds. Our analysis of the feasibility and benefit of cane toad quarantine is timely, given renewed  
313 emphasis on Australia's offshore islands as safe-havens to buffer biodiversity against cane toad  
314 impacts. Sixty-two Australian offshore islands designated as 'high conservation status' fall  
315 within the cane toad's predicted distribution; 21 of those have already been colonized by toads.  
316 Given our criteria (see Methods), we estimate the remaining value of toad quarantine across  
317 toad-free islands in northern Australia to be up to \$33 M AUD. This value is conservative for a

318 number of reasons. It is a reasonable expectation that as islands become home to increasing  
319 numbers of insurance populations or endangered species, the benefit of maintaining those islands  
320 as pest-free (measured as the cost of restoration) will increase. In addition, as toads establish  
321 onto an increasing number of these islands, those remaining toad-free will, by their scarcity  
322 alone, possess a greater economic and environmental value. While in reality it is unlikely that all  
323 islands without quarantine will be invaded, the benefit of quarantine within our dataset is held  
324 primarily in a few large islands (e.g. Melville Island, Table 1). These larger islands often have  
325 human settlements, competing management objectives (e.g., economic growth activities, multi-  
326 species quarantine) or more convoluted invasion pathways associated with anthropogenic  
327 activity. In short, quarantine needs to be carefully managed on these large islands. Eradication  
328 efforts for taxa other than toads have been successful on large islands (such as Santiago Island 5  
329 465km<sup>2</sup>; Cruz *et al.* 2009), but they require much greater planning, intersectional management,  
330 and investment in post-eradication surveillance and monitoring (Moore *et al.* 2010, Rout *et al.*  
331 2011, Carwardine *et al.* 2012). On these large islands, then, the value of prevention is very likely  
332 underestimated.

333 The vanguard of the cane toad invasion is currently sweeping across Western Australia at ~50  
334 km per annum, but recent research suggests that a waterless barrier between the Kimberley and  
335 the Pilbara could halt the toad invasion (Florance *et al.* 2011; Tingley *et al.* 2013; Southwell *et al.*  
336 2017). Applying our results to this management strategy revealed that the benefit of quarantine  
337 over such an area (\$34.3 – \$63.6 million) is roughly double the value of quarantine across all  
338 offshore islands combined (\$20.3 – \$38.5 million). The cost of quarantine in this case has been  
339 rigorously estimated at around \$5 million dollars over 50 years (Southwell *et al.* 2017), only a  
340 fraction of what we estimate it would cost to eradicate toads from this area.



341 Here we demonstrate the immense benefit of toad quarantine across northern Australia. We  
342 avoid arbitrary judgement and simply calculate the cost of eradication in the case of quarantine  
343 failure. We acknowledge that this value is undoubtedly a lower boundary on the true benefit, but  
344 valuing preventative management is important. It becomes more so as conservation actions  
345 increasingly rely on offshore islands and fenced areas as cost-effective avenues to protect  
346 biodiversity from the impacts of invasive species. Quarantine measures often protect against  
347 multiple potential invaders but our results suggest that even when considering a single species,  
348 the monetary value of quarantine can be substantial. Prevention, it seems, is worth more than we  
349 might naively guess, even with aphorisms to remind us.

#### 350 **Author's Contributions**

351 AS and BP contributed to all aspects of the manuscript. RT contributed to data collection and  
352 writing of the manuscript.

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363 **Data Accessibility**

364 Waterbody data are available upon request from Geographic Information Systems at the  
365 Department of Agriculture and Food, WA, and the Spatial Data Exchange section of the  
366 Department of water, WA. Cost data associated with our analyses can be found in the Supporting  
367 Information.

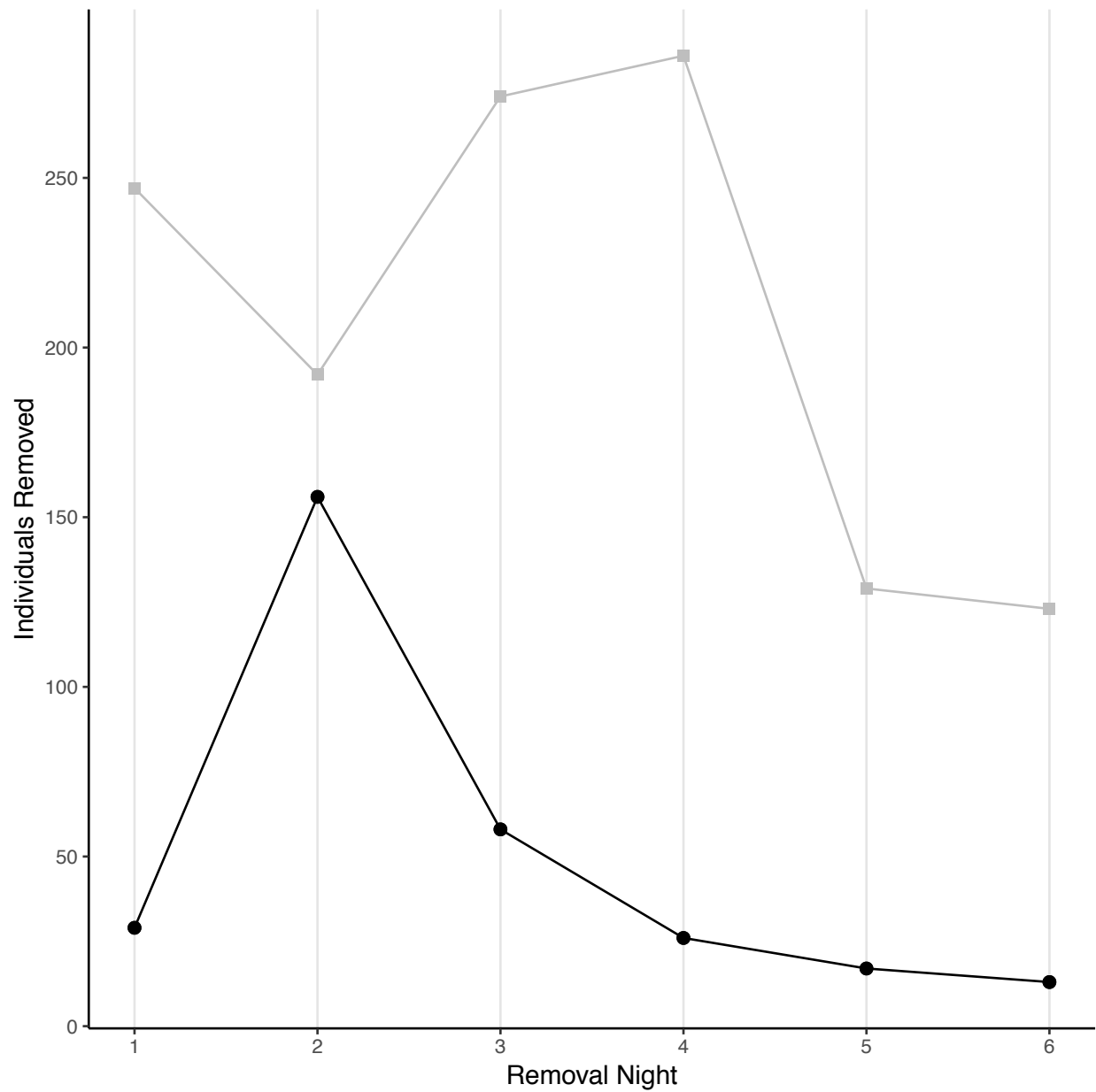
368 Table 1: Islands included in analyses from the top 100 islands prioritized by the Australian Commonwealth for conservation actions  
369 (Department of the Environment, Water, Heritage and the Arts (2009)). Estimates for the benefit of quarantine are in '000s (AUD).  
370 Mean benefit reports the cost of removal, averaging over costs calculated with the detection probabilities of each of our island  
371 systems.

<b>Jurisdiction</b>	<b>Island Name</b>	<b>Toads Present</b>	<b>Distance to mainland (km)</b>	<b>Area (km<sup>2</sup>)</b>	<b>Length of freshwater shoreline (km)</b>	<b>Mean benefit of quarantine (000s)</b>	<b>Lower Est.</b>	<b>Upper Est.</b>
New South Wales	Lord Howe Island	No	570	11	1	18	10	28
Western Australia	Barrow Island	No	56	139	21	373	200	580
	Bernier Island	No	38	171	2	36	19	55
	East Intercourse Island	No	5.5	51	2	36	19	55
	Faure Island	No	6.1	8	2	36	19	55
Queensland	Badu Island	Yes	90	53	10	178	95	276
	Bentineck Island	Yes	25	269	5	89	48	138
	Boigu Island	Yes	7.8	6	55	977	524	1519
	Darnley Island	Yes	70	195	0	18	10	28
	Dunk Island	Yes	4	170	1	18	10	28
	Goold Island	Yes	15	101	1	18	10	28
	Hammond Island	Yes	18	104	3	53	29	83
	Horn Island	Yes	16.7	396	8	142	76	221
	Macleay Island	Yes	3	16	0.7	12	7	19
	Magnetic Island	Yes	6.3	6	2	36	19	55
	Moa Island	Yes	52	72	21	373	200	580
	Moreton Island	Yes	20	7	54	959	514	1491
	Mornington Island	Yes	29	1662	102	1812	971	2817
	North Stradbroke Island	Yes	3.8	1001	105	1865	1000	2900
	Prince of Wales Island	Yes	16	148	27	480	257	746
Sweers Island	No	30	7	4	71	38	110	

Northern Territory	Bathurst Island	No	61	235	137	2434	1305	3783
	Centre Island	Yes	7.8	64	20	355	190	552
	Croker Island	No	3	11	152	2700	1447	4197
	Groote Eylandt	No	45	42	203	3606	1933	5606
	Marchinbar Island	No	21	5	59	1048	562	1629
	Melville Island	No	24	2	1054	18724	10036	29106
	North Island	Yes	28	13	3	53	29	83
	Peron Island	No	3.4	3	3	53	29	83
	Raragala Island	No	36	52	11	195	105	304
	Vanderlin Island	Yes	7	6	68	1208	647	1878
	West Island	Yes	4	576	30	533	286	828
	Yabooma Island	No	2.7	2	3	53	29	83

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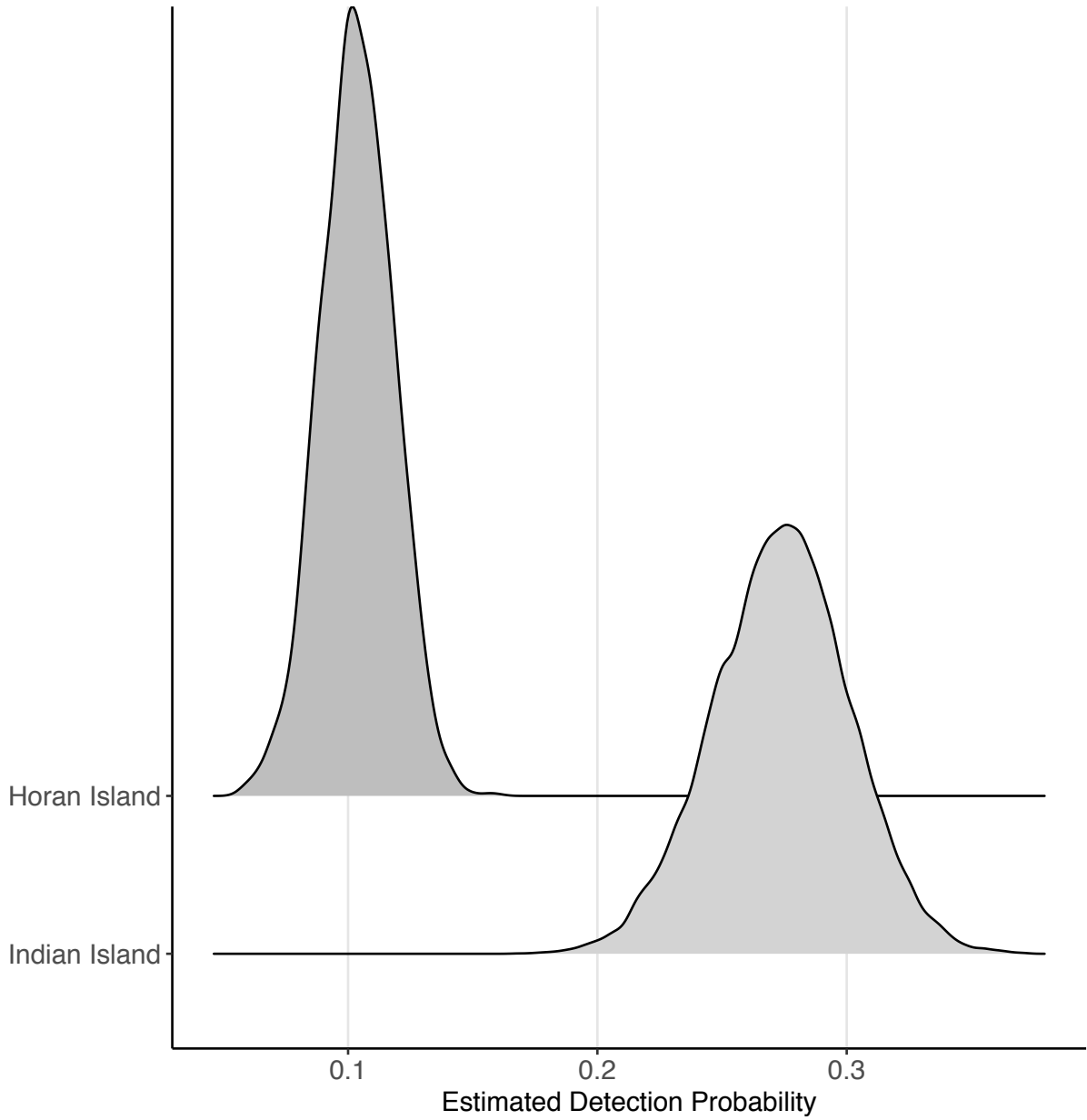
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374

375 Figure 1. Numbers of individual cane toads captured per night on Horan (gray)  
376 (black) Islands.

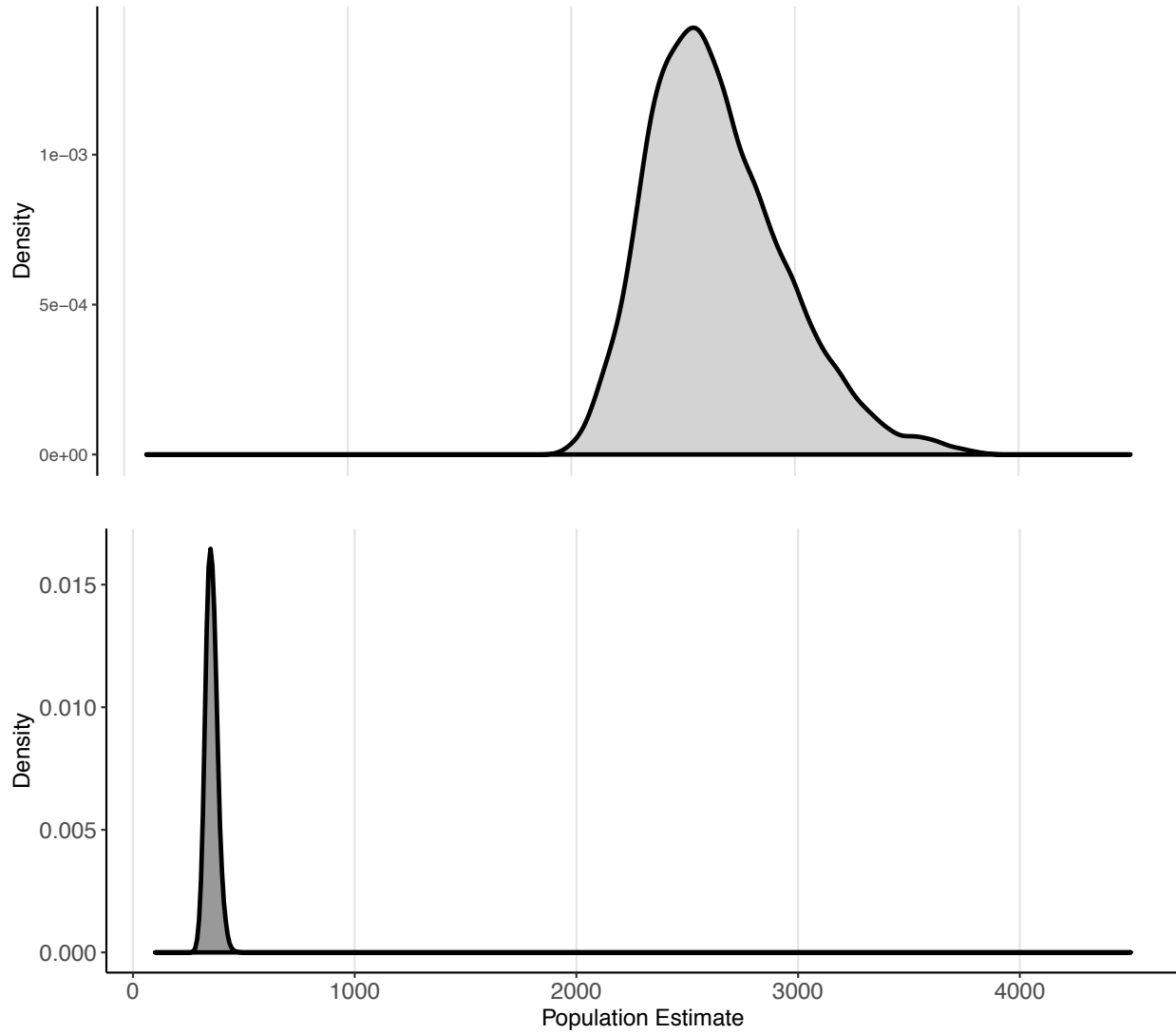
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378

379 Figure 2. Distributions of the estimated detection probabilities of cane toads on Horan and Indian  
380 Islands.

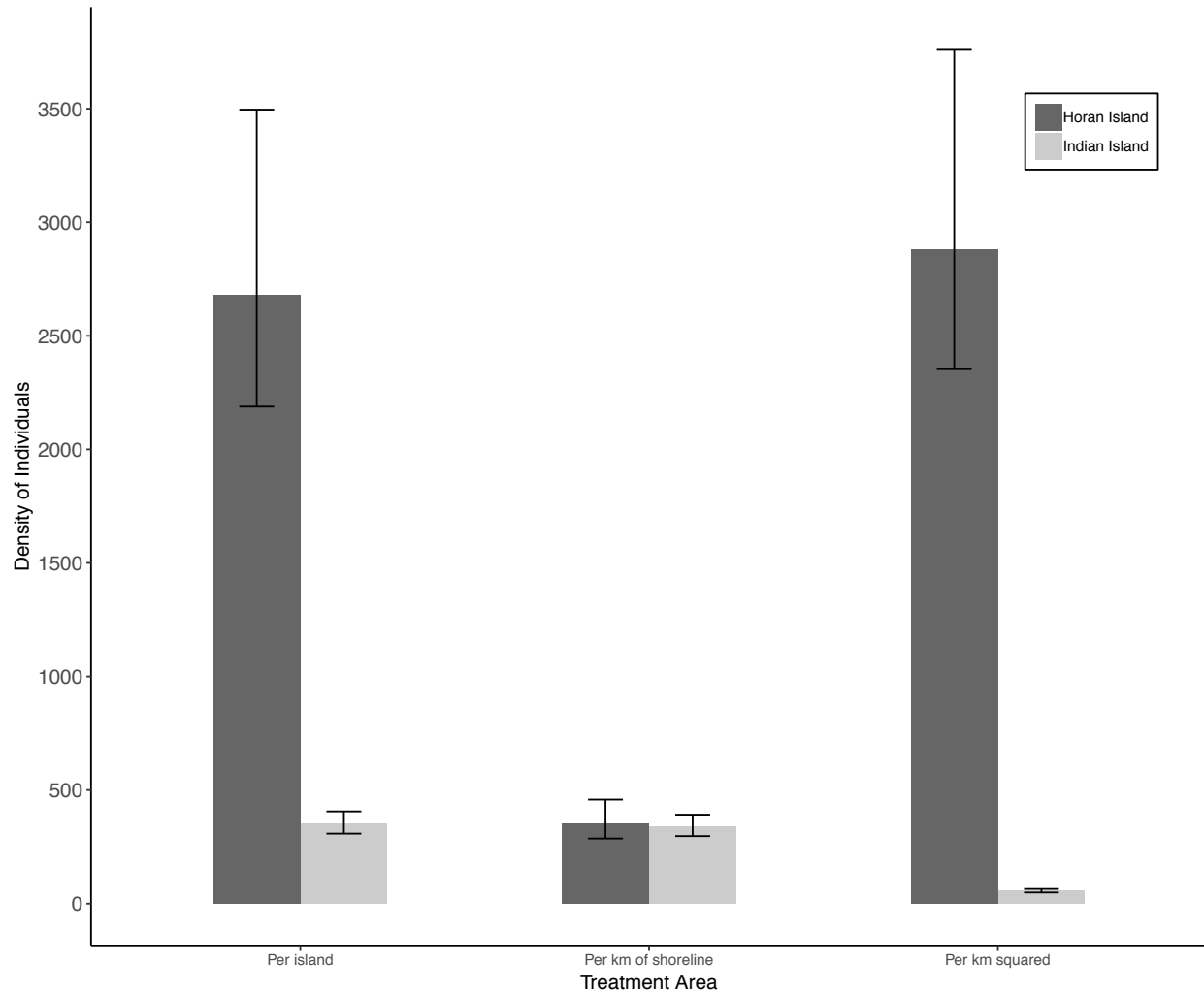
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382

383 Figure 3. Estimated distributions of cane toad population size estimates ( $N_0$ ) before removal  
384 efforts.

385



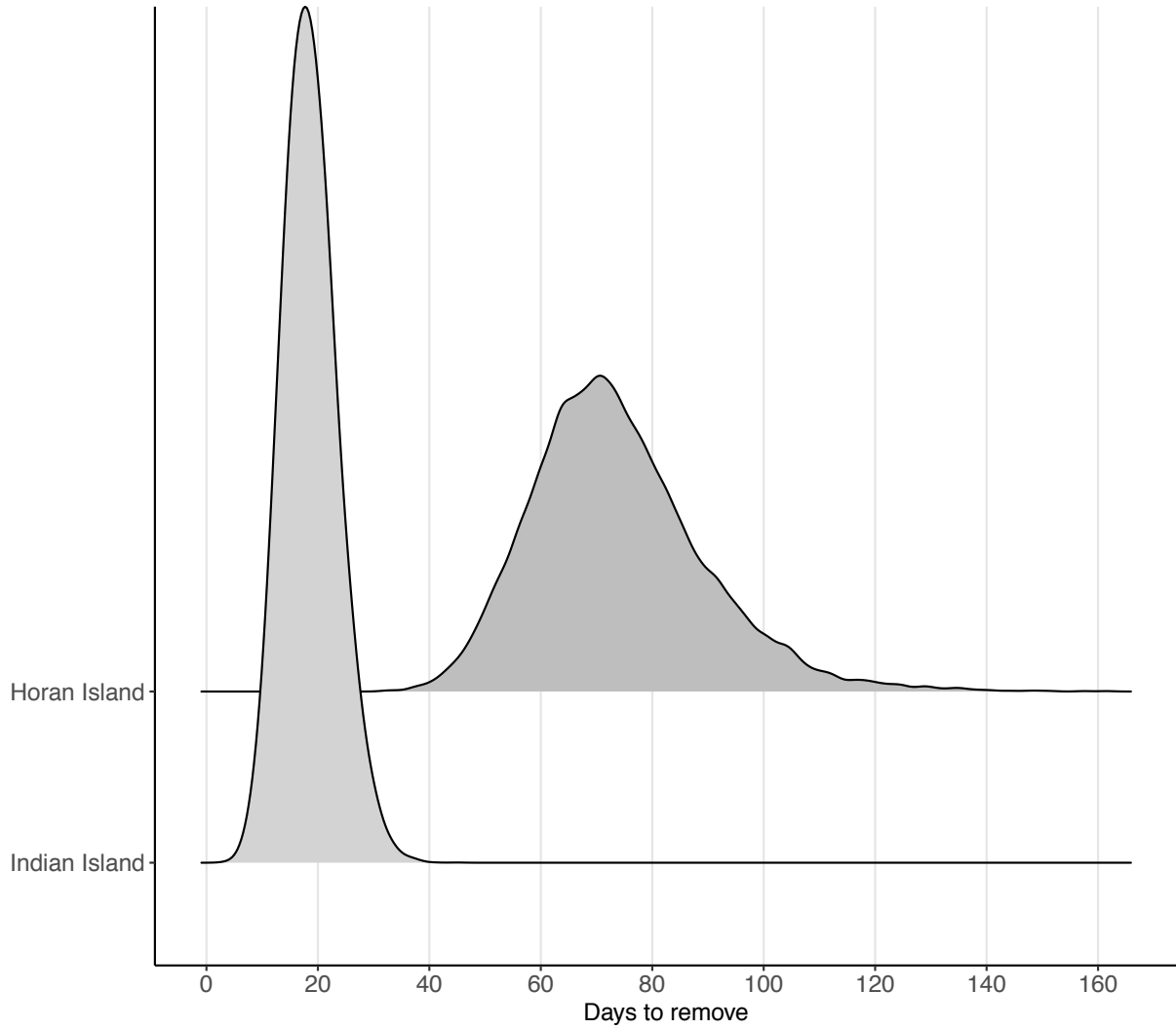
386

387 Figure 4. Estimated density of cane toads on each island using density calculated per island, per  
388 km, and per km<sup>2</sup>. There is clear concordance across islands when we calculate a linear density  
389 (per km).

390

391

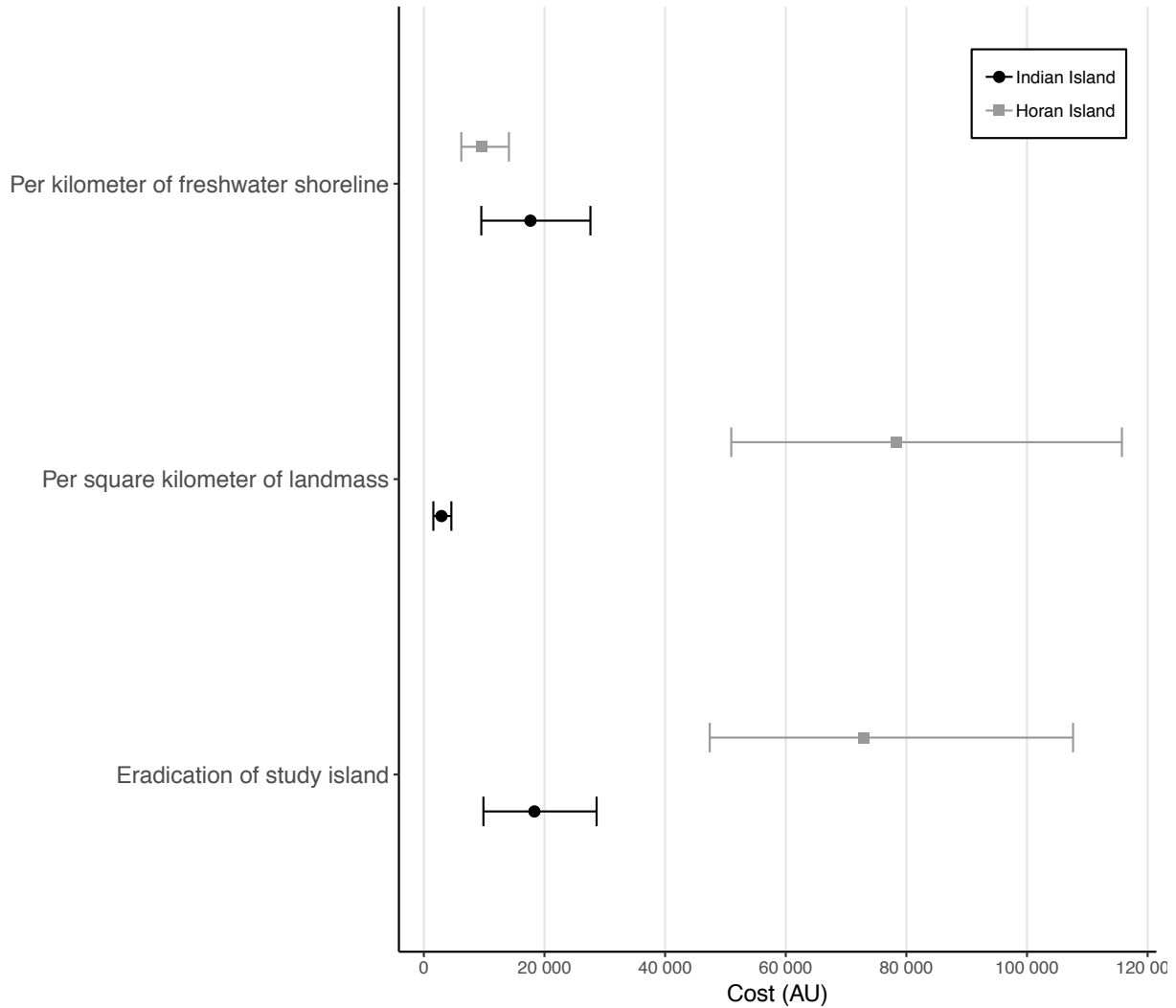




392

393 Figure 5. Distributions of the estimated numbers of days required to remove cane toads from  
394 Horan and Indian Islands.

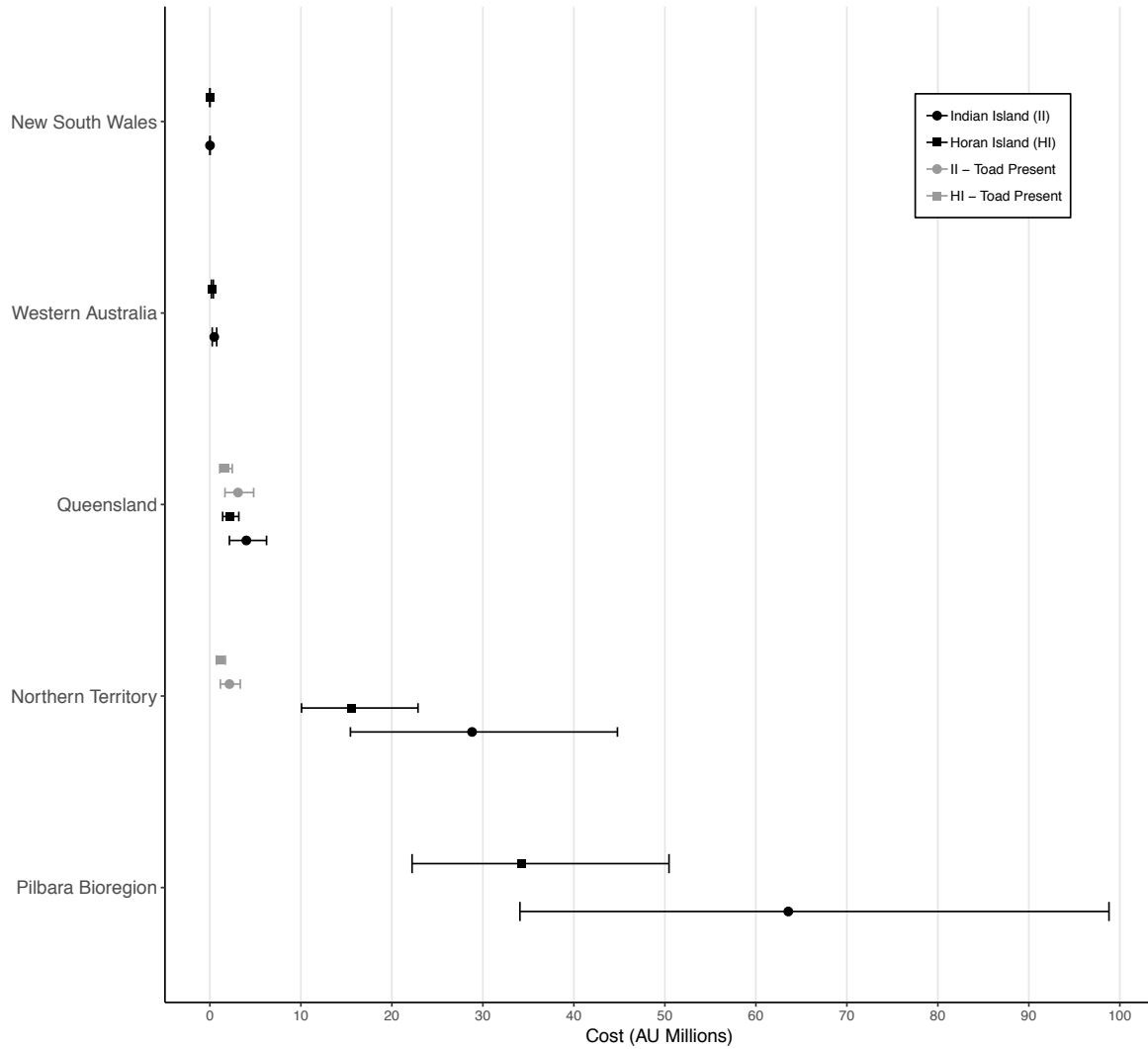
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397 Figure 6. Costs of eradication calculated per km of shoreline, per square kilometre, and per  
398 island. Again, we see the strong concordance in costs when we calculate them per km of  
399 shoreline.

400



401

402 Figure 7. Distribution of the benefit of cane toad quarantine across different jurisdictions within  
403 Australia. Toad present distributions denote areas where toads are known to occur, and represent  
404 the cost to remove toads. No islands in either New South Wales, Western Australia or the Pilbara  
405 Bioregion have confirmed toad presence.

406

407

408 **Supporting Information**

409 Supporting Table 1: Estimated costs of cane toad eradication derived from removal efforts in this  
410 study. All figures are in Australian Dollars (\$AU).

<b>Study</b>	<b>Cost</b>	<b>Mea</b>	<b>Low</b>	<b>Uppe</b>
Horan	Consumable	6 330	± 4	9 320
	Personnel	50	± 32	74
	Travel	16	± 10	24
Indian	Consumable	1 595	± 855	2 480
	Personnel	12	± 6	19
	Travel	4 142	± 2	6 438

411

412

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