


CONTRIBUTED PAPER

Detecting and protecting the threatened Kangaroo Island dunnart (*Sminthopsis fuliginosus aitkeni*)

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Reliable and cost-effective monitoring methods are a critical component of conservation management practices that work to prevent the extinction of threatened species. We evaluated the best means of monitoring the threatened Kangaroo Island dunnart (*Sminthopsis fuliginosus aitkeni*, hereafter KI dunnart). Variation in detection probability and cost was examined between four trapping methods. We then compared the occupancy of the KI dunnart in a 2017–2018 survey to a survey from 1999 to 2001. Across the 2017–2018 survey, the KI dunnart was detected at only five sites and was detected most frequently using camera traps on drift fence lines. The taxon was estimated to occupy 27% (95% confidence interval: 7–65%) of sites in the eucalypt woodlands of western KI. Of the methods that have successfully detected KI dunnarts, cameras on fence lines were the most cost-effective. Power analysis suggests that future monitoring surveys in spring and autumn must survey at least 55 sites to be capable of detecting a 60% decline in the KI dunnart population, and 26 sites to detect an 80% decline. Ongoing intensive monitoring is required to assess the population trajectory of the taxon and support its persistence on the island in the long term.

KEYWORDS

Australian mammals, camera trapping, fauna survey, methods, monitoring, occupancy modeling, pitfall trapping, power analysis, threatened species

1 | INTRODUCTION

Reliable detection methods are critical in ecological studies and wildlife management (Burton et al., 2015; Conn, Arthur, Bailey, & Singleton, 2006). While considerable effort may be invested in the study of rare and threatened species, the use of unreliable detection methods can produce erroneous estimates of conservation status, habitat requirements and population trends (Claridge, Mifsud, Dawson, & Saxon, 2005), thus

impeding effective conservation management (Lindenmayer & Likens, 2010; Woinarski, Garnett, Legge, & Lindenmayer, 2016).

For many animal groups, particularly small mammals, a wide variety of detection methods are now available, which vary in cost, effort required and efficacy (De Bondi, White, Stevens, & Cooke, 2010; Smith, Wilson, Moller, Murphy, & Pickerell, 2008; Welbourne, MacGregor, Paull, & Lindenmayer, 2015). Standard methods for detecting small

mammals include pitfall and Elliott (folding aluminum) traps, and camera traps. An increasing body of research has compared the effectiveness of these techniques, and others, for detecting different species (De Bondi et al., 2010; Garden, McAlpine, Possingham, & Jones, 2007; Vine et al., 2009). Camera traps are used increasingly in wildlife studies because they can be more cost effective than traditional methods and are often better able to detect cryptic species, run continuously for long periods, operate in extreme weather conditions, and are relatively uninvasive (Burton et al., 2015; De Bondi et al., 2010; Potter, 2017; Welbourne et al., 2015). However, while camera traps may be effective, sometimes other methods can be better at targeting particular species (Garden et al., 2007), if operating in certain environments such as dense vegetation (Swan, Di Stefano, Christie, Steel, & York, 2014), or monitoring species that are difficult to discern on camera (Welbourne et al., 2015).

Small variations in each of the methods discussed above can also have large effects on detectability (Friend, Smith, Mitchell, & Dickman, 1989; Ribeiro-Júnior, Rossi, Miranda, & Ávila-Pires, 2011; Thompson, Thompson, & Withers, 2005). For example, studies targeting the sandhill dunnart (*Sminthopsis psammophila*) and Butler's dunnart (*Sminthopsis butleri*) found that using pitfall traps that were twice the standard depth (640 mm vs. 320 mm) markedly increased the number of detections (Read, Ward, & Moseby, 2015; Ward, 2009). Camera trapping studies also indicate that detectability varies significantly with the camera model and set-up, and camera settings and deployment must be adjusted to specifically target the distinctive features and habits of the species in question (Claridge et al., 2005; Hohnen, Ashby, Tuft, & McGregor, 2013; Meek, Ballard, Vernes, & Fleming, 2015).

Notwithstanding a substantial previous sampling effort, and attention on the taxon because of its threatened status, there have been remarkably few records of the Kangaroo Island dunnart (*Sminthopsis fuliginosus aitkeni* [following the nomenclature of Jackson and Groves (2015)], hereafter referred to as the KI dunnart). This dearth of information may be because sampling techniques have been suboptimal and/or the taxon is restricted and rare. To help direct and prioritize its conservation management, more information on its distribution, habitat requirements, population size and trends is required. However, the collection of such information requires effective sampling. This taxon is a small (body mass ≤ 25 g) carnivorous dasyurid marsupial, restricted to KI (4,405 km²) in South Australia. The taxonomy of the species is contested, and it has been listed previously as *Sminthopsis aitkeni* (Kitchener, Stoddart, & Henry, 1984), and *Sminthopsis griseoventer aitkeni* (Kemper et al., 2011). It is listed nationally as Endangered (still under its former name of *Sminthopsis aitkeni*) by the *Environment Protection and Biodiversity Conservation Act 1999*. Since 1990, individuals have been reported on 31 occasions at nine sites on

western KI (Gates, 2001; Gates, 2011; Jones, Mooney, Ross, & Pisanu, 2010). The last extensive survey, by Gates (2001), occurred almost 20 years ago, between 1999 and 2001. A total of 22,655 trap nights (using pitfall and Elliott traps) across 44 sites on KI, yielded 22 individuals at six sites. Since then, the KI dunnart has been detected at two of those six sites, in 2009, and incidentally in three previously unsurveyed locations (Jones et al., 2010). In common with many other threatened Australian mammal species (Legge et al., 2018), there is currently no effective monitoring program for this species, at least in part because it has proven unusually challenging to sample.

In this study, we (a) compare the efficacy and cost of different methods to detect the KI dunnart, and (b) survey new locations as well as sites with historical records to assess the dunnart's current distribution. Specifically, these methods included: pitfall traps; Elliott traps; camera traps facing fence lines; and baited camera traps. We also compared capture success for other small mammals between Elliott and pitfall traps, and between three sizes of pitfall trap to determine the most effective way to live-trap dunnarts and other small mammals. KI dunnart detectability and occupancy were compared between our 2017–2018 survey and the 1999–2001 survey by Gates (2001). Estimates of occupancy were used to approximate the percentage of the island occupied by the dunnart. Our results help clarify aspects of the conservation status of the KI dunnart and the best means of monitoring this taxon in the future.

2 | METHODS

2.1 | Location and site selection

All survey sites were located in native vegetation on the western third of KI, within Flinders Chase National Park, Ravine des Casoars Wilderness Protected Area, and Kelly Hill Conservation Park (Figure 1 and Figure S1, Supporting information). This part of the island receives between 600 and 800 mm of rainfall annually (Bureau of Meteorology, 2017). In August–September 2017, 35 sites were surveyed. In March–April 2018, 34 of those sites were resurveyed along with eight additional sites (42 in total). Sites were stratified by postfire age classes: recently burnt (0–10 years postfire), middle age (10–20 years postfire) and long unburnt (>20 years postfire). Within these categories, most sites had an overstorey of KI mallee-ash (*Eucalyptus remota*), brown stringybark (*Eucalyptus baxteri*) or coastal white mallee (*Eucalyptus diversifolia*) (Table S1). The KI dunnart has been found in all of these vegetation associations (Gates, 2001). Sites were separated by at least 1.5 km to ensure they were independent, as other dunnart species have been known to make long distance movements of 1 km or more (Dickman, Predavec, & Downey, 1995).

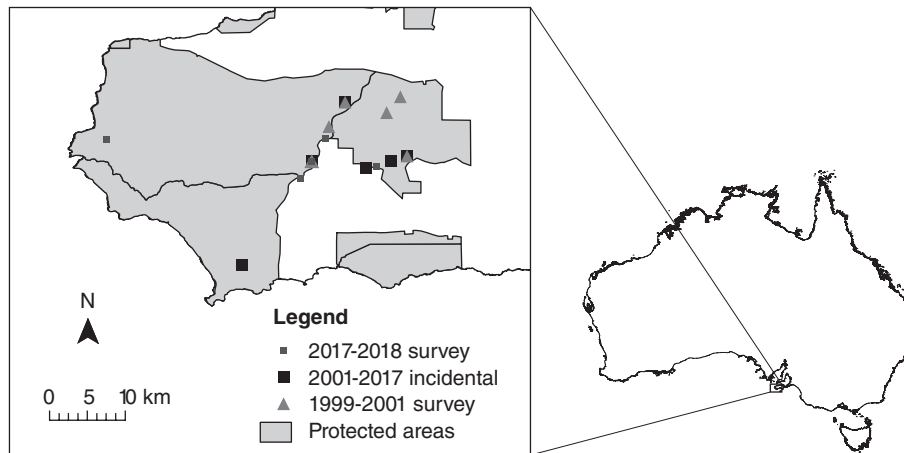


FIGURE 1 Kangaroo Island dunnart detections during the 1999–2001 survey, incidental records between 2001 and 2017, and detections from the 2017 to 2018 survey. Locations of all sites surveyed in both the 1999–2001 survey, and the 2017–2018 survey are available in Appendix S1

Study methods were approved by Charles Darwin University's Animal Ethics Committee (Permit Number A17009), and the South Australian Department for Environment, Water and Natural Resources (Scientific research permit E26661-1).

2.2 | Detection methods

Four different detection methods were used in the 2017 and 2018 surveys: (a) pitfall traps (with drift lines); (b) Elliott traps; (c) camera traps facing drift lines; and (d) baited camera traps. During each survey, a given site was initially trapped using Elliott traps (900 mm × 800 mm × 2,300 mm), pitfall traps with driftlines, and with cameras facing the driftlines (detection methods 1–3). Three 30 m long drift fences made of 60 cm high, 1 mm-thick plastic, held up by metal stakes, were dug into the ground. The branches of the drift fence met centrally, forming a “Y” shape (Figure S2). On each branch two pits were placed ~5 m from each end, resulting in a total of six pits per site. Three different pit sizes were tested: deep, wide pits (300 mm diameter, 650 mm depth), deep, narrow pits (150 mm diameter, 650 mm depth) and shallow, wide pits (300 mm diameter, 320 mm depth). Two infra-red Reconyx Hyperfire HC600 cameras (Reconyx, Holmen, WI) were deployed on either side of each driftline, facing towards the plastic fence (six per site). These cameras were unbaited and attached to stakes positioned ~1 m from the fence, 30 cm off the ground, and angled at 45° towards where the fence meets the ground, on the assumption that the drift lines would direct animals past the camera. Cameras were programmed to take three images per trigger, 1 s apart, with no minimum time delay between triggers. In addition to the pitfall and camera traps, 20 Elliott traps were placed around the periphery of the site (Figure S2). Elliott traps were baited in the late afternoon with a mixture of peanut butter, oats and wet cat food and were checked and closed at dawn.

All sites were trapped for 3 days and nights, with Elliott traps open at night and pitfalls open at night and during the day. If it rained at night, pitfall and Elliott traps were closed but the cameras were left running, and as multiple days might pass before the pits could be opened again (so that they were available for three nights total) fence line cameras were open for between three and seven nights. For all mammals captured, a small portion of hair was clipped off the rump with a pair of fine scissors as a mark of recapture within a given survey period. All captured animals, including mammals, reptiles and amphibians, were identified to species level. Reptiles were marked using permanent marker on their underbelly as a mark of recapture, but amphibians were not marked.

After a site had been trapped for 2 days and 3 nights, the pits were closed, and the fences removed. Three baited Reconyx Hyperfire HC600 or PC800 cameras (Reconyx, Holmen, WI) were then set up and left at the site for 10–20 days. At each site the baited cameras were positioned ~30 m apart. Each camera was attached to a stake 30 cm off the ground facing a bait station positioned 1 m away. This consisted of a tea strainer containing five mealworms, attached to a stake, with some peanut butter and oat mixture crumbled alongside. Inside the tea strainer mealworms had access to a small amount of oatmeal which helped them survive the 10-day period. Cameras were programmed to take three images per trigger, 1 s apart, with no minimum time delay between triggers. The entire 2017–2018 survey resulted in a total of 11,445 trap nights, of which 1,386 were pitfall trap nights (462 nights per pitfall trap type), 4,620 were Elliott trap nights, 2,268 were fence line camera trap nights, and 3,171 were baited camera trap nights.

Data from the 1999 to 2001 KI dunnart survey (Gates, 2001) were compared with the 2017–2018 surveys to explore changes in KI dunnart detectability and occupancy. Briefly, the 1999–2001 survey consisted of 44 sites trapped across KI between November 1999 and April 2001, with a total of 22,645 trap nights (13,317 Elliott and 8,941 pitfall

trap nights). Sites were distributed in natural vegetation in national parks, conservation areas and on some areas of private land across KI (Figure S1). These sites were stratified by the dominant overstorey plants, namely *E. remota*, *E. diversifolia*, and *E. baxteri*. Each survey site consisted of two subsites placed ~100 m apart. Each subsite consisted of four pitfall traps arranged in a square and placed 20 m apart with 5 m of drift fence extending from each side of the pitfall traps (Figure S2). Pitfall traps were mostly narrow and deep (150 mm diameter, 500 mm depth) but some wide and shallow buckets were also used (395 mm diameter, 295 mm depth). Also, between 8 and 15 Elliott traps were deployed at each site but these data were not included in our analysis due to the lack of information on trap effort per trapping occasion. Sites were trapped for 3–7 nights at a time, and some on multiple occasions, the most resulting in a site being trapped for 67 nights (Gates, 2001).

2.3 | Statistical analysis

2.3.1 | Live trapping data

Variation in capture frequency of commonly caught small mammal species in the 2017–2018 sampling was first compared between Elliott and pitfall traps and then between the three sizes of pitfall trap. This was done using generalized linear models (GLMs), within the quasipoisson error family, in R version 3.5.3 (R Development Core Team, 2018).

2.3.2 | Occupancy models

We used single-season occupancy models to examine variation in detection probability (i.e., probability of detecting a species, if present at a site) of the KI dunnart between the four different survey methods used in the 2017–2018 survey (pitfall traps, Elliott traps, cameras on fence lines and baited camera traps) (Hohnen & Gates, 2018). Data from the 1999 to 2001 survey were not included in this part of the analysis as we aimed to compare methods that were conducted at the same sites, during the same time period, and could not be confounded by change in occupancy between surveys. The probability of detection was modeled as a function of two observation covariates: “detection method” (pitfall traps, Elliott traps, cameras on drift fencelines, and baited cameras), and “lure effectiveness” (expected to decline at a constant rate at baited camera traps). Models were fitted using the ‘occu’ function in the “unmarked” package in R version 3.5.3 (R Development Core Team, 2018). The best model was identified using Akaike's information criterion (AIC), with models within two AIC units of the top model considered competitive.

We then used the same type of model to examine how occupancy of the KI dunnart varied between the 2017–2018 and the 1999–2001 surveys (Gates, 2001). The variables “detection method” and “season” were included as covariates. “lure effectiveness” was not included as a covariate, as it was only relevant to the 2017–2018 survey and was a poor

predictor of dunnart detectability in the previous analysis (see details below).

Occupancy results were used to approximate the amount of the island where the KI dunnart might still persist. Despite extensive surveys the KI dunnart has not been recorded below the 800 mm isohyet in the last 39 years (Gates, 2011). However, above the 800 mm isohyet they have been found in all major vegetation associations including *E. remota*, *E. baxteri*, *E. diversifolia*, and *Eucalyptus cladocalyx* (Gates, 2001). The site-based estimate of KI dunnart occupancy (from the 2017 to 2018 survey data), was therefore multiplied by the area comprised of eucalypt woodland within the 800 mm isohyet (where all survey sites were situated [1,301 km²]). This value was then divided by the total area of KI (4,405 km²), to reach an approximate percentage estimate of the amount of the island occupied.

2.3.3 | Cost analysis

We examined cost variation between methods that successfully detected KI dunnarts in the 2017–2018 survey (baited and fence line camera trapping) and the 2001 survey (pitfall trapping). For each method, the cost required to survey a single site was calculated for the number of nights required to reach a detection probability of 95%. This was calculated from the nightly detection probability of each method produced by the dynamic occupancy model in which method was included as the only variable. To keep the results broadly applicable we included the cost parameters of “vehicle” and “labor” but did not include food and accommodation. We assumed that the surveyor was living in a town 80 km from the sites, and therefore drove this distance to set them up and, if necessary for the given method, check them each day. We assumed pitfall traps would be checked once in the morning and once in the late afternoon, requiring one trip to the sites per day. Cameras on fence lines and baited cameras were treated as though there were checked only once halfway through the trapping period. Equipment costs were calculated per site, and it was assumed that the cost was spread evenly over a minimum of 20 surveys, so 5% of the total equipment cost was included in the cost calculations (De Bondi et al., 2010) (see Table S2 for cost formulae used in this analysis). Labor costs were based on the hourly rate of a senior research assistant at Charles Darwin University which is \$95.60 (\$51.74/hr × 1.32 [on-costs] × 1.40 [over-heads]), with the same rate charged both for field hours and image processing. Vehicle running costs were based on Charles Darwin University's vehicle reimbursement charge, which is 0.66 c/km. All costs in the manuscript are stated in Australian dollars, with a currency conversion of AUD \$1 = USD \$0.72 (from November 27, 2018). The number of images recorded per night was calculated from the results of the 2017–2018 survey by dividing the total number of images (detected using a given method) by the number of sites and then by averaging the number of nights the site was

open for (Table S2). Based on previous experience we estimated that 10,000 images could be processed per day.

2.3.4 | Power analysis

We used estimates of detectability and occupancy from the 2017 to 2018 survey in a power analysis used to determine the number of sites required to pick up changes in dunnart occupancy. Power analyses are designed to determine if the study design in place has a good chance of producing statistically significant results with a biologically relevant effect size. In the analysis the cost of both Type I and Type II error was deemed to be equal, so we used Type I error level of 0.2 and Type II error level of 0.8. Two situations were examined, the first where the survey was conducted in spring and autumn and the second where the survey was conducted only in spring. In both situations the survey method was assumed to be camera traps on fence lines (as it had the highest detection probability of any method). Autumn was not examined separately as there were too few detections for the models to converge and produce an accurate measure of detectability and occupancy. For both situations we set the number of repeat visits as the number of nights required to reach a cumulative detection probability of 95% (30 nights in spring and autumn and 17 nights in spring). We examined the number of sites required to detect a change in occupancy of 30, 50, and 80% in accordance with aspects of the IUCN criteria for listing as vulnerable, endangered, and critically endangered (IUCN Species Survival Commission, 2012). All analyses were conducted using R code provided by Guillera-Aroita and Lahoz-Monfort (2012).

3 | RESULTS

3.1 | Detections of the KI dunnart

In the 2017–2018 survey the KI dunnart was detected using camera traps at five sites (Figure 1) on seven occasions, but no individuals were caught in pitfall traps or Elliott traps. Camera traps on fence lines detected the KI dunnart at four sites, and baited camera traps detected individuals at two sites.

3.2 | Live trapping

Although the KI dunnart was not live-trapped, we live-trapped seven other mammal species (Tables S3 and S4). Pitfall trapping resulted in the capture of 7 mammal, 10 reptile, and 5 amphibian species, and Elliott trapping resulted in the capture of 4 mammal species. Mammals were captured more frequently in Elliott traps, driven largely by the bush rat (*Rattus fuscipes*) and the house mouse (*Mus musculus*) (Figures S3 and S4). Both little pygmy possums (*Cercartetus lepidus*), and western pygmy possums (*Cercartetus concinnus*) were caught significantly more often in pits

(Figure S4 and Table S4). In total a greater number of mammal species was caught in pitfall traps compared with Elliott traps (seven and four respectively). In contrast at each site in each trapping period, Elliott traps caught a greater number of mammal species significantly more frequently than pitfall traps (Figure S4). Captures of southern brown bandicoots (*Isoodon obesulus*), common brushtail possums (*Trichosurus vulpecula*) and swamp rats (*R. lutreolus*) were too low to be included in the models. The number of reptile individuals and species captured were significantly higher in pitfall traps than Elliott traps (Figure S4).

Western pygmy possums and bush rats were the only mammals caught frequently enough in pitfall traps to compare pitfall trap sizes. Capture frequency of both species was highest in wide, deep pits, and lowest in narrow, deep pits (Figure S5 and Table S4). Mammal diversity and abundance varied between pitfall trap sizes and was highest in wide deep pits. In contrast, amphibian diversity was higher in wide shallow pits, but reptile diversity and abundance did not vary significantly between the pitfall trap sizes.

3.3 | Camera trapping

In total, cameras on fence lines recorded 31,158 images, with identifiable animals triggering the cameras on 349 occasions in August–September 2017 and 569 occasions in March–April 2018 (i.e., 0.4 animal images per trap night). This method resulted in the detection of 13 mammal, 2 reptile and 9 bird species, but no amphibians (Table S3). Baited cameras recorded 314,049 images, with identifiable animals triggering the cameras on 4,446 occasions in August–September 2017, and on 6,197 occasions in March–April 2018 (i.e., 3.4 animal images per trap night), and resulted in the detection of 13 mammal, 1 amphibian, 5 reptile and 18 bird species (Table S3).

3.4 | Occupancy–detection models: Detection method

The model that best described the probability of KI dunnart detection in the 2017–2018 survey included “detection method” as a predictor variable. The second best model, the null model (i.e., containing no predictor variables), was within two AIC units of the best model, and the model containing the variable “lure effectiveness” was inferior to the null model (Table S5). The model averaged coefficient estimates indicated that detectability was highest using camera traps on fence lines, followed by baited camera traps, and was lowest using pitfall and Elliott traps (Figure S6).

Differences in occupancy were compared between the 2017–2018 and 1999–2001 surveys using single season occupancy models. The best model contained the variables “season” and “method,” but two other models, each containing a single variable, “season” or “method,” were competitive with the best model (within two AIC units). Model-averaged coefficient estimates indicate that occupancy of dunnarts

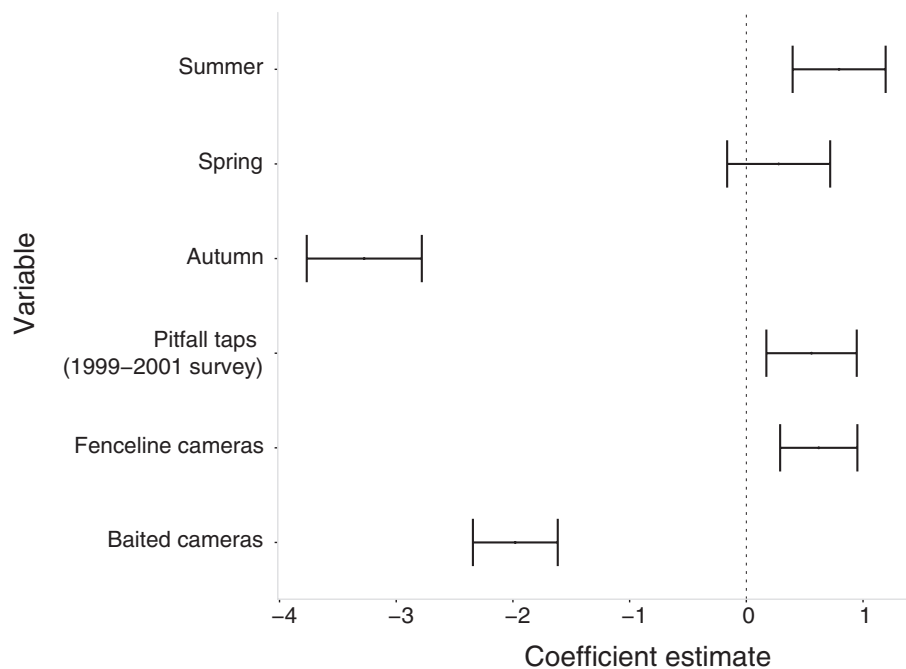


FIGURE 2 Model averaged coefficient estimates with 95% confidence intervals of the single season occupancy model fitted to the Kangaroo Island dunnart survey data from 1999 to 2001 and 2017 to 2018. For clarity, the methods Elliott trapping, and pitfall trapping method 1 (from the 2018 to 2018 survey), and the season winter were omitted from the figure as no dunnarts were detected during those seasons/using those methods

was highest in summer, followed by spring, autumn, and then winter (Figure 2). Coefficient estimates were also highest for fence line cameras from the 2017 to 2018 survey, followed by pitfall methods from the 1999 to 2001 survey, and baited camera traps from the 2017 to 2018 survey. Model-averaged estimates of occupancy were virtually unchanged between the two surveys (0.26 and 0.27, respectively); however, the confidence intervals of these estimates were very large, indicating considerable uncertainty (Figure S7). Capture rates (compared between different methods) were comparable with previous surveys, with a rate of 1.8 individuals caught per 1,000 nights on fence line cameras in the current survey and 1.5 individuals per 1,000 nights in pitfall traps in the 1999–2001 survey (Table 1). Based on these results, if dunnarts occupy 27% of eucalypt woodlands within the 800 mm isohyet on KI, then they likely occupy approximately 351 km² ($0.27 \times 1,301 \text{ km}^2$), and 7.9%

of the island as a whole (351 km² occupied/4405 km² total island area).

Three methods were successful at detecting dunnarts: fence line cameras and baited cameras in 2017–2018 and pitfall traps in 1999–2001. To reach a cumulative nightly detection probability of 95% using the methods from the 1999 to 2001 survey, the models predict that a site needs to be trapped for 51 nights. However, using the array of six camera traps on fence lines from the 2017 to 2018 survey, sites would need to be trapped for 29 nights, approximately half the amount of time required by using the 2001 sampling approach (Figure 3). Finally, an array of three baited camera traps would require 125 trap nights to reach a detection probability of 95%.

3.4.1 | Cost analysis

The cost of each method was a mixture of the equipment costs and the labor costs. Overall the cheapest method (when trapped for enough nights to reach a 95% detection probability) was camera traps on fencelines, at \$ 2,497 per site. Baited camera traps on fence lines were the next cheapest method, costing \$ 4,985 (Figure 4 and Table S6). The majority of the cost associated with both camera trapping methods was attributable to labor, and costs were higher for baited camera traps (compared to fence lines) as they attracted more nontarget species and therefore much higher image processing time. The most expensive method was pitfall trapping, which (to trap for 51 days and reach a 95% detection probability) was estimated to cost \$ 28,945 (Table S6). While the equipment costs were low, a considerable amount of labor is required to dig the sites in and then recheck them twice daily to release and process the animals (Figure 4 and Table S6).

TABLE 1 Trap success (number of captures/number of trap nights) for the Kangaroo Island dunnart from the 1999 to 2001 survey by Gates (2001) and 2017–2018 survey, using different methods

Survey period	Detection method	Pitfall trap nights	Captures	Trap success (captures per 1,000 nights)
1999–2001	Pitfall	13,714	21	1.5
1999–2001	Elliott	8,941	1	0.1
2017–2018	Pitfall	1,386	0	0.0
2017–2018	Elliott	4,620	0	0.0
2017–2018	Fence line camera traps	2,268	4	1.8
2017–2018	Baited camera traps	3,171	2	0.6

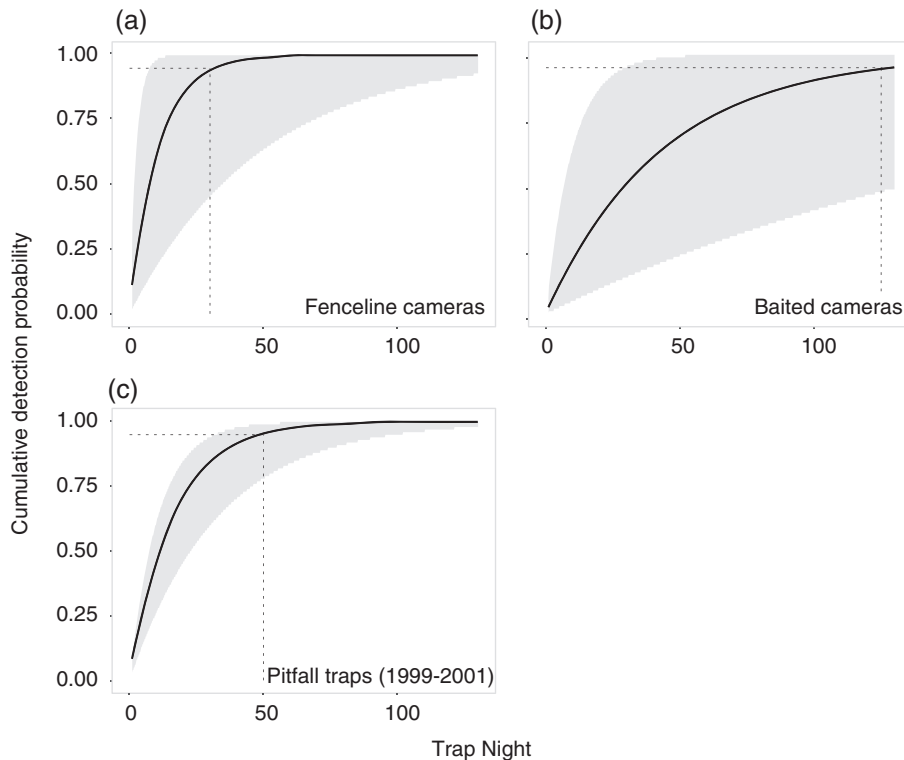


FIGURE 3 Cumulative nightly detection probability of Kangaroo Island dunnarts using: (a) pitfall traps (1999–2001 survey data), (b) camera traps facing fence lines (2017–2018 survey data), (c) baited camera traps (2017–2018 survey data), taking 51, 30, and 125 days (respectively) to reach a 95% detection probability

3.4.2 | Power analysis

As the occupancy and detection probability of the KI dunnart varied between the seasons, so did the number of sites required to detect a given change in occupancy in a given season. To detect a change in occupancy of 80%, 26 sites would have to be surveyed in both spring and autumn, and 69 sites if the survey was conducted only in spring. To detect a 50% change in occupancy 84 sites would have to be surveyed in spring and autumn or 221 sites if the survey was conducted

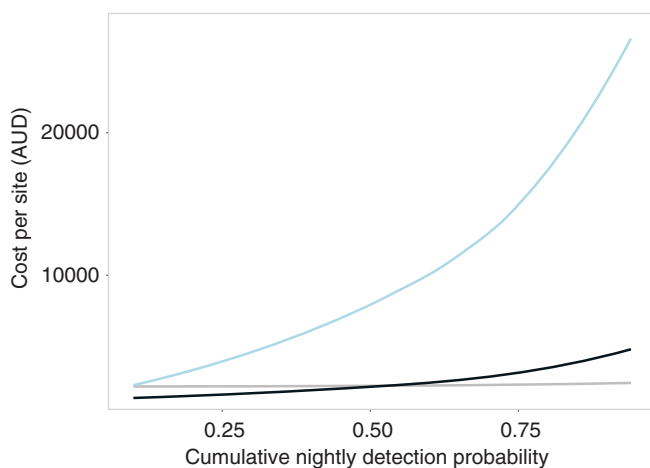


FIGURE 4 The relationship between cost and cumulative nightly detection probability for the Kangaroo Island dunnart of trapping a site using pitfall traps (blue line), camera traps on fence lines (grey line), and baited camera traps (black line) which take respectively, 51, 30, and 125 days to reach detection probability of 95%

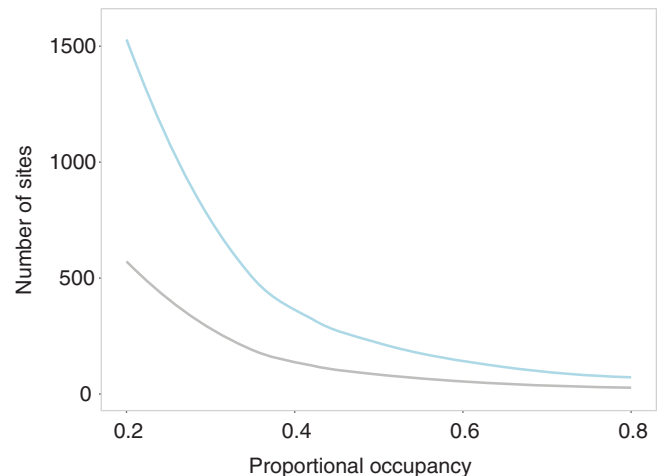


FIGURE 5 The number of sites required to detect a change in occupancy of Kangaroo Island dunnarts, if surveyed in spring and autumn (grey line) and if only surveyed in spring (blue line)

only in spring. Detecting a 30% decline in the dunnart population would be very difficult, as 260 sites would have to be surveyed in both seasons, and 692 sites if the survey was conducted in spring alone (Figure 5 and Table S6).

4 | DISCUSSION

Our results suggest that the KI dunnart currently occupies about 27% (95% confidence interval: 7–65%) of the eucalypt

woodlands of western KI, and therefore is restricted to 8% of KI's total area (8% of 4,405 km² = 352 km²). We are unable to conclude, with any reasonable level of confidence, whether there has been a reduction in site occupancy by the KI dunnart in the last two decades. However, the small geographic range of the dunnart (most likely <500 km²) makes it vulnerable to disturbance events such as large wildfires, as well as ongoing attrition by threats such as predation by feral cats. Ongoing monitoring of the KI dunnart at a large array of permanent monitoring sites would provide a clearer understanding of trends in abundance and the area of occupancy. Camera traps on drift fence are the most effective method of detecting the KI dunnart, although we note that an ideal protocol would be to extend their duration of use at a site from the 3 nights used here to ca. 30 nights. Surveying 55 sites in spring and autumn would be sufficient to detect a moderate to large change in occupancy (60%), and surveying 26 sites in both seasons would be sufficient to detect only a drastic change in occupancy (80%).

KI has long been considered a stronghold for a number of vertebrate species that are now rare on mainland South Australia, such as Rosenberg's goanna (*Varanus rosenbergi*), pygmy copperhead (*Austrelaps labialis*), tammar wallaby (*Notamacropus eugenii*), bush stone-curlew (*Burhinus grallarius*), glossy black-cockatoo (*Calyptorhynchus lathamii*), and southern brown bandicoot (*I. obesulus*) (Gates & Paton, 2005; Pepper, 1996; Rismiller, McKelvey, & Green, 2010). However, declines of some species have occurred, such as Rosenberg's goanna (P. Rismiller, personal communication, June 3, 2018), the heath mouse (*Pseudomys shortridgei*), which has not been sighted on the island since 1967 (Kemper, Medlin, & Bachmann, 2010), and earlier still the local extinction of the spotted-tailed quoll (*Dasyurus maculatus*) (Haouchar et al., 2014). It is clear from these declines that while some species have persisted on KI longer than on the nearby mainland (perhaps due to lack of threats such as red foxes *Vulpes vulpes* and European rabbits *Oryctolagus cuniculus*), some of the threats that have caused declines elsewhere are still present on the island (such as land clearing, feral cats and the plant pathogen *Phytophthora cinnamomi*). While the data from this survey do not suggest the KI dunnart has declined significantly since 2001, the taxon appears to have a small area of occupancy (<500 km²). The low number of detections at sites where it was found, as well as the lack of detections at sites with historical records, suggest that the KI dunnart is at best elusive and at worst in low numbers in areas where it persists. Further targeted monitoring is required to better understand its population trajectory.

The aims of the current study were to resample sites with historic dunnart records and to find the taxon in new locations, but not to completely replicate the 1999–2001 survey. There are potential problems in comparing the recent survey with the earlier survey: first, the sampling methods varied between the two surveys; and second, all but six of the

survey sites (those with positive dunnart detections in the earlier survey) differed. A common feature of both surveys was that most sites were located on the western side of the island (31 of the 44 sites in 1999–2001 and all sites in 2017–2018). Both surveys focused on the same vegetation types, with 71% of sites in 2017–2018, and 85% of sites in 1999–2001, in vegetation dominated by an overstorey of *E. remota*, *E. baxteri* or *E. diversifolia*. These overstorey vegetation types are the most common on KI (Table S1). Gates (2001) found the KI dunnart in each of these three associations as well as open woodland dominated by *E. cladocalyx*.

Of the four detection methods tested, camera traps on fence lines were the most effective at detecting the KI dunnart. The dunnart was also detected on baited camera traps but at a much lower frequency. This adds to findings from other studies that camera traps are often better able to detect some rare and cryptic animal species, in comparison to traditional live trapping methods (Burton et al., 2015; Karanth & Nichols, 1998; Meek et al., 2015). The drift fence lines acted to funnel dunnarts towards the cameras, potentially making them more likely to be detected. The other benefit, specifically for detection of the dunnart, was that cameras on fence lines had lower image processing time compared to the baited camera traps as there were fewer triggers by nontarget species (i.e., 6,197 triggers of baited cameras, compared to 569 triggers of fence line cameras in the March–April 2018 survey).

In situations where a species is attracted to a particular bait, baited camera traps are likely to be more successful than fence line cameras at detecting individuals. In this study, the KI dunnart did not appear to be attracted to bait, as no individuals were detected entering Elliott traps, and only two were detected on baited camera traps, one of which occurred after the bait canister had been removed by a common brushtail possum (*T. vulpecula*). Potentially, dunnarts were excluded by other species that were more strongly attracted to the bait such as common brushtail possums and native bush rats. Baited cameras and Elliott traps have been found to be less effective than pitfall traps at detecting other dunnart species including the common dunnart (*S. murina*), Butler's dunnart (*S. butleri*), and sandhill dunnart (*S. psammophila*) (De Bondi et al., 2010; Potter, 2017; Read et al., 2015). However, baited camera traps were quicker to set up and did detect more species than any other detection method (37 species), substantially more than fence line cameras (24 species) and pitfall and Elliott traps (22 and 4 species, respectively). This difference was partly driven by the large number of bird species detected by baited cameras (18 species), which was double the number of bird species detected on fence lines (9 species). Overall, baited cameras appear more effective at detecting a wide range of species, and therefore may be more useful in broad scale monitoring,

rather than monitoring that targets a particular taxon such as the KI dunnart.

While camera trapping on drift fence lines might be the most effective means of detecting the KI dunnart, our results suggest that if animals did need to be live caught (e.g., for mark–recapture studies, health surveillance or for captive breeding), pitfall trapping is likely to be most effective. During the 1999–2001 survey 21 of the 22 captured KI dunnarts were trapped in pitfall traps, suggesting that Elliott traps are not effective at detecting dunnarts (Gates, 2001). Small mammal species live trapped in the 2017–2018 survey (*R. fuscipes*, *R. lutreolus*, *M. musculus*, *C. concinnus*, and *C. lepidus*) were caught most frequently in wide deep pits (300 mm diameter, 650 mm depth). Other studies suggest that small mammals including *Rattus* and *Sminthopsis* (dunnart) species can jump out of wide, shallow pits (diameter > 300 mm, depth < 400 mm), such as 20 L buckets, commonly used in general fauna surveys in Australia (Read et al., 2015; Ribeiro-Júnior et al., 2011; Thompson & Thompson, 2010). The use of wide deep pits has certainly increased capture success of other dunnart species elsewhere in Australia (Read et al., 2015; Ward, 2009). However, the results of the 1999–2001 survey indicate that narrow deep pits (150 mm diameter, 600 mm depth) can also be used to effectively trap KI dunnarts. Interestingly, amphibians were caught at highest frequencies in wide shallow pits but reptiles did not show strong selection for any pitfall trap size. While animals are potentially more vulnerable to predation in pitfall traps, there was no evidence that predation occurred in pitfalls in this study.

When considering the cost required to reach 95% detection probability, fence line camera traps were the cheapest detection method (\$ 2,497 per site) followed by baited camera traps (\$ 4,985 per site), and pitfall traps (\$ 28,945 per site). Also, while baited camera sites need half the number of cameras than fence line sites do, they also need to be left out for four times as long to reach a detection probability of 95% (125 nights compared to 30 nights). Using cameras on fence lines, twice as many sites can be surveyed in the same amount of time with the same number of cameras. Despite low equipment costs, pitfall trapping was not competitive (in terms of overall cost) with either of the other methods as the large amount of labor involved meant that, after 51 nights (needed to reach 95% detection probability), the cost was over 11 times higher than camera traps on fence lines. In this situation, camera traps on fence lines are the most cost-effective option.

Overall, this study adds to a growing body of evidence suggesting that camera trapping is an especially effective method for detecting cryptic ground dwelling small mammals (De Bondi et al., 2010; Hohnen et al., 2013). Furthermore, the results suggest that taking time to consider camera trap set up, and making sure it effectively targets the species in question, is critical (Claridge et al., 2005; Harley, Holland, Hradsky, & Antrobus, 2014; Srbek-Araujo & Chiarello,

2013). The use of inaccurate detection methods can result in an erroneous representation of a species' current distribution and population trajectory, and therefore the application of ineffective or even harmful management actions. The outcome of such actions could be particularly detrimental if the species is already rare and/or threatened. The approach suggested by this study involving camera traps on drift fence lines could be an accurate and cost-effective method for detecting other cryptic small mammal species, particularly those that are not readily attracted to bait.

A recent study identified the KI dunnart as one of the 20 Australian mammals at greatest risk of extinction (Geyle et al., 2018). The results of our study suggest the KI dunnart occupies less than one third of the remnant eucalypt woodlands on western KI, and less than 8% of KI's total area. Overall, the results of this study indicate that camera traps on fence lines are the most accurate and cost-effective method for detecting the KI dunnart. The use of such methods on surveys in the future could significantly increase our knowledge of the distribution and ecology of this species as well as other cryptic and threatened small mammals elsewhere. Although we were unable to ascertain, with any reasonable degree of confidence, whether rates of site-occupancy by the KI dunnart have changed over the last two decades, it is concerning that: (a) there are no ongoing, targeted monitoring programs in place to give us a clear picture of the population trajectory; and (b) there are no insurance populations off the island or in captivity (Gates, 2011). Addressing these deficits, along with improving our understanding of its habitat requirements and ecology, will help ensure the long-term survival of the KI dunnart.

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CONFLICTS OF INTEREST

The authors acknowledge no conflicts of interest in the production of this manuscript.

AUTHOR CONTRIBUTIONS

Rosemary Hohnen designed the project, collected and analyzed the data, and wrote the manuscript. Brett Murphy was

involved in the project design, data analysis and manuscript editing. Jody Gates was involved in the project design, data collection and manuscript editing. Sarah Legge, Chris Dickman, and John Woinarski were involved in the project design and manuscript editing. All authors have given approval of the version to be published and have agreed to be accountable for all aspects of the work.

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