This is the peer reviewed version of the following article: Moyses, J., Hradsky, B., Tuft, K., Moseby, K., Golding, N. and Wintle, B. (2020), Factors influencing the residency of bettongs using one-way gates to exit a fenced reserve. *Austral Ecology,* Volume 45, Issue 5, Pp 858-871; which has been published in final form at **https://doi.org/10.1111/aec.12898**.

Abstract

Understanding the conditions under which small native Australian mammals can persist in the presence of introduced predators remains a key challenge to conservation ecologists. Bettong-specific *one-way gates* were used at a predator-free reserve in South Australia to allow the burrowing bettong (*Bettongia lesueur*) – a small potoroid, listed as ‘vulnerable’ nationally - to disperse out of the reserve. We conducted a field experiment to explore the conditions affecting residence time of bettongs that left the reserve. We monitored bettong and mammalian predator activity outside the fence using track surveys across 18 sites over two seasons. We examined the effect of supplementary feeding as a strategy for increasing residence time, as well as the influence of predator presence and habitat quality, using linear mixed models. Bettong activity was positively associated with supplementary feeding, midstorey vegetation cover and shelter availability. After gates were closed, bettong activity near gates declined to almost zero the following weeks, likely either due to death from predation or movement away from the sites. To a small extent, mammalian predators were more likely to be present at sites with high bettong activity. Further research on conditions to support persistence of burrowing bettongs and other small mammals, including understanding minimum necessary predator control effort, is required before successful establishment of populations outside of fences can occur.

key words

burrowing bettong, translocation, augmented feeding, invasive predator, reintroduction, threatened mammal, feral cat, arid zone

Introduction

In Australia, 32 terrestrial mammals have become extinct since European settlement and a large proportion of persisting mammal species occupy less than 20% of their former range (Short & Smith 1994; Woinarski *et al.* 2015; Waller *et al.* 2017). A disproportionate number of extinctions and declines have occurred among ‘critical weight range’ (CWR) mammals, weighing 35 – 5,500 g (Burbidge & McKenzie 1989). These species are particularly susceptible to predation by feral cats (*Felis catis*) and European foxes (*Vulpes Vulpes),* and their loss has negatively impacted ecosystem function (James & Eldridge 2007).

Conservation on islands or fenced reserves free of introduced mammalian predators is currently the only option to protect several Australian CWR species and prevent further extinctions (Legge *et al.* 2018). Yet, while fenced reserves have produced insurance populations of 38 predator-susceptible species, developing and maintaining reserves comes at a cost. Reserves are limited in area and capacity to expand. Contained mammals lose predator avoidance instincts unless predators are also introduced, or predator-avoidance behaviours are instilled through training (Blumstein *et al.* 2002; West *et al.* 2018). Notably, there is also risk of overpopulation because populations may not regulate within closed systems where the full guild of native predators are absent (Slotow et al. 2005; Moseby *et al.* 2018a). Fences also inhibit the dispersal and movement of free-ranging animals outside the reserve (Slotow *et al.* 2005; Hayward *et al.* 2014). These challenges within predator-free areas have the potential to result in counterproductive outcomes for both conservation and animal welfare.

There are currently few solutions to minimising effects of overpopulation within predator-free reserves. Culling is widely seen as socially unacceptable and counter to conservation goals (Moseby *et al.* 2018a), options for translocation on the scale required may not be available, and regulation by mammalian predators within fenced reserves is not yet fully established. Reintroducing threatened mammals outside predator-free areas therefore is desirable as a solution to overpopulation, but more importantly to establish viable, self-sustaining populations (IUCN 2012) and return species’ ecological functions to the landscape (James & Eldridge 2007). However, medium-sized mammal reintroductions beyond fences in Australia have had mixed outcomes (Moseby *et al.* 2011). The major challenge in reintroducing medium-sized mammals outside of predator-free areas is managing the threat of predation by cats, foxes and potentially native species (e.g., dingoes *Canis dingo*) (Short *et al.* 1992; Moseby *et al.* 2011; Hardman *et al.* 2016). Further research is required to determine whether successful reintroductions are possible with additional management actions (Short *et al.* 1992; Moseby 2011a).

Actions to promote survival after release from a fenced sanctuary could include increasing predator control, improving habitat conditions (including increased provision of shelter) and soft-release methods. In some circumstances, introduced predators such as feral cats and foxes can be controlled through poison baiting, shooting and trapping (Algar *et al.* 2013); however, effectiveness of control depends on environmental conditions, location, density of predators and reinvasion rate (Moseby 2011b; Christensen *et al.* 2013). Alternatively, soft-release methods can be used to increase the chances of establishment (e.g. Bright & Morris 1994, Short & Turner 2000; Tuberville *et al.* 2005; Mitchell *et al.* 2011), Soft-release methods such as the provision of supplementary food, shelter or acclimatisation pens aim to reduce the negative effects of sudden release by enhancing site fidelity, familiarising animals with release site food and shelter, and minimising stress and panic-dispersal in early stages of reintroduction (Moseby *et al.* 2014). The provision of supplementary food and water have been tested previously with varying results (e.g. Bright & Morris 1994; Rickett *et al.* 2013; Bannister *et al.* 2016).

This study sought to test availability of shelter and supplementary food as measures for improving reintroduction outcomes in a release of CWR mammals conducted to prevent poor conservation and animal welfare outcomes from overpopulation within a fenced reserve.

Arid Recovery (hereafter 'AR') is an ecosystem restoration initiative comprising a 123 km^2 reserve. Burrowing bettongs (hereafter 'bettongs'), which formerly persisted across a large proportion of southern Australia are now listed as 'vulnerable' nationally after having become extinct on the mainland (Department of Environment, 2019). Bettongs were reintroduced to a predator-free section of AR Reserve in 1999 (Moseby *et al.* 2011), and within ten years became overabundant (Moseby *et al.* 2018a). The overabundance has negatively impacted vegetation, with likely flow-on negative impacts to the population of greater stick-nest rats (*Leporillus conditor*) in the reserve, which rely on the same food resources as bettongs (Linley *et al.* 2017; Moseby *et al.* 2018a).

A strategy tested at AR is bettong-specific one-way gates: small rectangular walk-ways with a vertical Perspex flap allowing animals to push their way out of the fence (Crisp & Moseby 2010). The current model of one-way gates has proven effective in allowing bettongs to disperse outside the fence, with very low rates of non-target species using the gates and no incursions of introduced species (Butler *et al.* 2018).

Bettongs' social behaviour and use of central warrens make them highly vulnerable to predation by introduced species such as cats and foxes (Christensen & Burrows 1994). There have been two trials to reintroduce bettongs from AR into unfenced, predator-controlled areas. Both were ultimately unsuccessful, mainly due to predation, but also the rapid dispersal of released animals and/or drought conditions (Moseby *et al.* 2011; Bannister *et al.* 2016), although there was some evidence that supplementary feeding improved short-term survival of bettongs close to feeders (Bannister *et al.* 2016). In contrast, in an experiment within a fenced section of AR, a population of bettongs has been able to establish and breed while co-existing with a controlled number of feral cats over a four year period (Moseby *et al.* 2018b).

The release of bettongs through one-way gates at AR during 2017 and 2018 provided an opportunity to build on learnings from the two previous bettong reintroductions to sites beyond the fence at AR. Following a study by Bannister *et al.* (2016), which detailed poor survival of translocated bettongs, our study aimed to clarify the potential of supplementary feeding to enhance bettong residence time on dunes outside the AR fence close to one-way gates, and investigate how the presence of predators and variation in habitat influenced the residence time. The experiment was designed to add value to AR's bettong management program, which uses one-way gates to facilitate dispersal outside the reserve. The results are expected to inform future outside-of-fence reintroductions of the species. We predicted that bettong activity outside the fence would be high at one-way gate sites (and decline less rapidly over time) where: (i) greater numbers of bettongs dispersed through one-way gates, (ii) supplementary food was provided and (iii) vegetation cover and shelter availability were high. We predicted that predators would be attracted to the bettongs and so would be more likely to occur at sites where more bettong tracks were recorded.

Methods

Study area

The study took place at AR (30˚29'S, 136˚53'E), 20 km north of Roxby Downs, South Australia. Feral cats, red foxes, dingoes and European rabbits (*Oryctolagus cuniculus*) are excluded from a 60 km^2 area within the reserve (Moseby and Read, 2006). Burrowing bettongs were the second species - after greater stick nest rats – to be reintroduced to the reserve in 1999 (Finlayson & Moseby 2004). Other species reintroduced are the greater bilby (*Macrotis lagotis*), western barred bandicoot (*Perameles bougainville*) and recently, western quoll (*Dasyurus geoffroii*).

The landscape is dominated by east-west trending dunes with a sparse cover of tall shrubs and trees including wattle (*Acacia ligulata*), hopbush (*Dodonaea viscosa*), mulga (*Acacia aneura*) and native pine (*Callitris glaucophylla*), separated by inter-dunal swales dominated by low chenopods: saltbush (*Atriplex vesicaria*) and bluebush (*Maireana astroticha*). Ephemeral canegrass (*Eragrostis australasica*) swamp habitat is also found on the East boundary and open claypans on the West boundary of AR. The climate is hot and dry with an average summer maximum temperature of 38˚C, average winter minimum temperature of 4˚C and average rainfall of 143 mm per year (Bureau of Meteorology 2019). A hot summer with below average rainfall led to drought conditions at AR in 2018 (Bureau of Meteorology 2019).

Study species

The bettong is a CWR mammal (average weight 1300g, height 500 – 700 mm) in the Potoroidae family (Short & Turner 2000). Bettongs formerly lived across a large proportion of western and southern mainland Australia but are currently confined to three natural populations on islands off the coast of Western Australia, and translocated populations at AR in South Australia, Scotia Sanctuary in New South Wales and Yookamurra Sanctuary in South Australia and Faure Island in Western Australia (Woinarski *et al.* 2015; Kanowski *et al.* 2018).

Bettongs are nocturnal, feeding on shrub roots, stems, leaves, bulbs, insects and fungi (Bice & Moseby 2008). They function as 'ecological engineers' as their digging and foraging promotes nutrient enrichment for understory plants (Noble *et al.* 2007; James *et al.* 2009; Chapman 2015).

Experimental Design

We collected field data in two survey seasons: Spring 2017 and Autumn 2018.

Study sites were split along the West boundary and East boundary of the reserve (Fig. 1a). Gates were installed on the Western and Eastern boundaries of the reserve, due to the location of habitat outside the reserve, and the high abundance of bettongs in these areas. All gates were south of the dingo fence where dingoes are controlled. However, the close proximity to the dingo fence (< 10km) meant that some dingoes were present in the study area. Gates on the Western boundary were installed prior to the commencement of this study in 2016 and gates on the Eastern boundary installed in 2017. Due to East-West trending dunes, the location of gates also allowed monitoring of animal tracks to be within a similar distance from the fence across sites. In 2017, we monitored 11 sites on the West boundary and seven on the East boundary. In 2018, we monitored nine of the 11 sites on the West boundary and all seven East boundary sites.

Each site comprised a one-way gate and an associated east-west dune running approximately perpendicular to the AR fence (Fig. 1b). At each West boundary site, we monitored bettong and predator tracks along a 300-m transect along the ridge of the dune closest to the one-way gate – starting five metres from the fence. On the East boundary, transect lengths varied from 80 - 300 m (mean length 270 m) because dune habitat at some sites rapidly transitioned to salt bush swale or ephemeral canegrass swamp, making it difficult to observe animal tracks. Sites were 300 – 1000 m apart.

In each season, we provided supplementary food at ten sites (of 18 surveyed in 2017 and of 16 surveyed in 2018), including sites on the East and West boundaries. We placed feeders 100 m from one-way gates and within 50 m of track transects underneath tall shrubs. Feeders comprised a 50 cm long PVC pipe (diameter = 10 cm) fixed above a 20 cm diameter plastic feeding tray and attached by wire to two two-metre pickets hammered into the soil. During the treatment stage we ensured feeders contained 1500 g of feed (rolled oats) each evening.

In each season, we monitored sites across two phases (Fig. 1c):

- Phase 1. This 'gates open' phase comprised firstly a non-feeding stage where one-way gates were open, but no supplementary food was provided outside the gates (two weeks in 2017; one week in 2018), followed by a treatment stage when supplementary feeding (hereafter 'feeding') was introduced to feeding sites (four weeks in 2017; two weeks – East boundary, and one week – West boundary 2018). This phase allowed a baseline to be established for the bettong track counts at each site before one-way gates were closed.
- Phase 2 ('gates closed' phase) comprised closing one-way gates and continued feeding at feeding sites for the remainder of the season (three weeks in 2017; five weeks – East boundary, and three weeks – West boundary 2018).

We extended Phase 2 in 2018 to extend monitoring of bettong residence time for an extra two weeks after closing gates. The lengths of monitoring stages were varied between the East and West boundaries in 2018 to accommodate essential fence maintenance works on the eastern boundary at the start of this survey season.One-way gates monitoring

Figure 1. (a) Locations of one-way gate sites on the East and West boundaries of Arid Recovery, South Australia. (b) Magnified view of site 15 on the East boundary. (c) Survey timeline for 2017 and 2018 seasons. The experiment was conducted in two Phases. Phase 1 included a period of monitoring in which one-way gates were open and food placed daily – including an initial non-feeding stage at all sites (yellow) and an experimental treatment stage where supplementary feeding was introduced at feeding sites (green). Phase 2 involved a period of monitoring in which one-way gates were closed and treatments continued (orange).

We checked open one-way gates daily and put ~20g of peanut butter outside the gates to attract bettongs through the gates. We used camera-traps to determine the number of individuals moving through one-way gates. One-way gates were open for 41 nights each on the West and East boundaries in 2017, and 12 nights on the West boundary and 22 nights on East boundary in 2018 (Fig. 1c). We deployed two *Reconyx HC600* cameras at each gate while gates were open, one camera facing the inside part of the gate, and one facing the exit point, with the following settings: high sensitivity, *RapidFire*, no delay between triggers, one image per trigger (Butler *et al.* 2018).

We separated exit images from non-exit images by comparing inside and outside camera trap images. An exit was determined by an image of a bettong entering a gate followed by an image of the bettong outside of the fence within 1 minute (Fig. A1.1 Supp. mat). An exit was only counted if the inside and outside cameras were triggered. We collated exit images and recorded the number of exits through each gate on each night.

Track monitoring

The afternoon prior to surveying transects, we smoothed a track along the dune at each site by dragging a one-metre-wide rubber mat with rope along the length of the transect (Fig. A1.2). We surveyed transects between one and three hours after sunrise the following morning to allow good visibility of tracks. We recorded the number of bettong tracks – counting one track each time an animal crossed onto and off the transect (Moseby *et al.* 2011). Predators (cats, foxes and dingoes) were recorded as present or absent for each transect (Table A2.3).

We repeated each transect survey every three to four days throughout the survey seasons. All transects were surveyed 15 times in 2017; East boundary transects were surveyed 19 times and West boundary transects 10 times in 2018.

Habitat surveys

At each site, we surveyed vegetation along two perpendicular 100-m line transects, centred on the dune 100 m from the start of the track transect (Fig. 1b). We used a 'touchpole' method to record the presence of vegetation contacting a 1.5-m pole in two height classes: 'midstorey' (0.3 – 1.5 m) and 'canopy' (>1.5 m) (adapted from D'Antonio *et al.* 2011). The method involves placing a pole marked at fixed heights along a transect and recording vegetation presence or absence (or vegetation type or species) at the heights of interest. Midstorey vegetation was considered to indicate potential protection from terrestrial predators, while canopy vegetation potentially indicated protection from aerial predators. Touches were recorded at 0.5 m intervals along each line transect (400 points in total). We then calculated the percentage of touches for each height class, derived the median, and categorised the vegetation cover as 'low' (less than the median; 10.5% for midstorey vegetation and 2.75% for canopy vegetation) or 'high' (equal to or greater than the median).

We recorded available shelter at each site by counting rabbit warrens for 300-m along each dune on which track surveys were conducted (Table A2.2). To ensure the entire width of the dune was searched, three people walked along the dune together 30 – 50 m apart and marked warren locations with a *Garmin etrex* 10 GPS. Warrens of any number of entrances (usually between two and five) and warrens appearing to be actively or recently used by rabbits were considered. There were no warrens observed in the swale habitat between dune study sites.

Predator control

Cat and fox control occurred outside the fence before and during both field seasons out to a buffer of 10 km from the reserve, including trapping, poison baiting at 5 baits/km of track, and spotlight shooting. Nineteen cats and one fox were removed on 14 shooting nights before and during the Spring 2017 field season, and 100 poison (0.04 g/kg sodium fluoroacetate '1080') semi-dried kangaroo meat baits were distributed by hand on vehicle tracks within five km of the perimeter in August 2017.

Before and during the Autumn 2018 field season, five cats and one fox were killed on six shooting nights. Eight hundred semi-dried kangaroo meat baits were laid in March and April 2018 in a larger buffer up to 10 km around the reserve. Nineteen cats were caught in foothold and cage traps set around the reserve on 297 trap nights within two weeks prior to the Autumn 2018 field season.

Statistical analysis

Bettong activity

To test our predictions about the drivers of bettong residence time, we characterised the relationship between bettong track count after the gates were closed (Phase 2) and the independent variables listed in Table 1 using generalised linear mixed models with a Poisson link function, appropriate when the dependent variable is a count. Models were fitted in R v3.3.3 (R Core Team 2013) using package *lme4* (Bates *et al.* 2014). We standardised continuous independent variables by dividing raw data values by maximum values. A random intercept structure was included to account for the nestedness of the survey design (repeat surveys of sites grouped along East and West boundaries within each survey season) (Table 1). An observation-level random effect was also included to account for overdispersion of count data cf. the Poisson distribution; equivalent to a Poisson log-normal observation model (Harrison 2014). To meet the model assumptions of homogeneity of residuals it was necessary to include an interaction term between the season (2017 vs. 2018) and side (east vs. west), and to include a quadratic term for the effect of day since gate opening on track counts.

Predator occurrence

To test our *a priori* hypotheses about drivers of predator occurrence, we characterised the relationship between predator presence throughout the monitoring period, including after gates were opened (Phases 1 and 2) and the independent variables listed in Table 1, using generalised linear mixed models with binomial error distributions and a logit-link function, appropriate when the dependent variable is binary. We determined predator presence by the presence of cat, fox and/or dingo tracks on a transect. Again, a random intercept structure was included to account for the nestedness of the survey design. Presence of cat, fox and dingo tracks were analysed together.

Model selection

For each set of predictions, we used function *dredge* in R package *MuMIn* (Barton 2013) to compare the best model subsets of the full model set (Table A3.1). In the bettong activity model we always retained random effects and the season/side interaction in the subset models. We also allowed fitting of two-way interactions of interest, for example an interaction term for days and feeding treatment. The null models for bettong activity and predator occurrence comprised only the random effects and season/side interaction as predictor variables. For both models, variable selection was based on Akaike's Information Criterion adjusted for small sample size (AICc) which measures performance based on a trade-off between model fit and model complexity (Akaike 1973; Cavanaugh 1997). The top-ranked bettong activity model included all variables in the top dredged models (>2 AICc of the first model), with the observation-level random effect. We used the top-ranked bettong activity and predator occurrence models to create fitted response plots of individual predictors.

We ensured that top-ranked models met statistical regression modelling assumptions of homogenous, independent, and appropriately-dispersed residuals by analysing randomised quantile residuals with the R package *DHARma* (Hartig 2018), visual checks, simulation tests, and Durbin-Watson tests for temporal autocorrelation (Fig A3.1; A3.2; A3.3).

Results

Bettongs

In total, we recorded 1223 bettongs exits through the one-way gates in spring 2017 and 349 bettongs exits in autumn 2018. Average exits per gate per night were 0.93 on the West and 2.69 on the East in 2017, and 0.38 per gate per night on the West and 2.37 on the East in 2018 (Table A2.1). No bettongs were observed regaining access to the reserve through the one-way gates once they had exited.

In both survey seasons, the number of bettong tracks on transects increased immediately after gates were opened but started to decline at some sites before gates were closed. Tracks declined after one-way gates were closed, particularly in 2018 (Fig. A2.3; Fig. 2). There was strong consistency among the six top-ranked models of bettong activity (track count) after gates were closed (Table 2). The fixed effects of each of the six models explained approximately 90% of the variation in track counts (Table 2). The number of bettong tracks was substantially higher at sites where food was provided and lower at sites where predators were present (Table 2; Fig. 2; Fig. 3a). The interaction between food and predator presence had a positive effect on daily track counts which reinforces that predators tended to be attracted to sites of high bettong activity (Table 2; Fig. 3a). Shelter availability was strongly positively associated with daily track counts (Table 2; Fig. 3a), and mid-storey shrub cover and canopy vegetation cover were weakly positively associated with daily track count (Table 2; Fig. 3a). At all sites, the daily count of bettong tracks declined with time since gate closure (Table 2; Fig. 2). Where food was provided, the number of tracks declined more rapidly (Fig. 2). In summary, while food increased initial bettong activity, the number of tracks at sites with high initial bettong activity (feeding sites) tended decline fastest. The association between high activity sites and predator presence provides some support to the hypothesis that predators, rather than dispersal were the main agents of decline following gate closure, and that high bettong activity tends to attract predators in.

Predators

Despite the effort to control predators prior to opening one-way gates, feral cats and foxes persisted in the area. Trapping rates of feral cats along the fence line were particularly high prior to gates opening in 2018 (19 cats caught in two weeks). We recorded predator tracks in 81% of transect surveys in 2017 compared to 67% in 2018. Feral cats predominated, with relatively few records of foxes and dingoes (Table 3). While conducting track surveys, we found evidence of wedge-tailed eagles (*Aquila audax*), feral cats and dingoes preying or scavenging on bettongs. In Spring 2017, 33 bettong carcasses were found nearby the East boundary and three nearby the West boundary. In Autumn 2018, 24 bettong carcasses were found on the East boundary. Twice each, feral cats and dingos were detected on camera traps by gates on the East boundary in 2018. Wedge-tailed eagles were observed killing bettongs that had recently exited one-way gates and were yet to establish warrens. Once warrens were established, the rate of wedge-tailed eagle predation appeared to decline. Feral cat tracks were observed close to transects and showed evidence of bettong predation, including one instance of a cat predating a bettong close to a feeder.

Our top-ranked model of predator presence over the entire monitoring period (i.e. while gates were open and after they were closed), showed that survey season (2017/2018) had the strongest effect, with predators more likely to be detected in 2018 than 2017 (Table 2; Fig. 3b; Fig. 4). To a small extent, predators were also more likely to be recorded as the number of bettong tracks increased (Fig. 4), averaging 70% presence on any given transect when bettongs were abundant. Supplementary food was not selected in the top-ranked predator presence model, however, the positive relationship between feeding and bettongs and the positive relationship between bettongs and predators indicates that anything that creates a high bettong density, including feeding will likely attract predators.

Figure 2: Fitted response plots for variables included in top-ranked bettong activity model, showing the response of bettong track counts per day per transect to food and predator presence in a), b), c) and d) 2017 and e), f), g) and h) in 2018 at East boundary sites. Line shows mean of fitted response with 95% confidence limits shaded. Plotted points are recorded values.

Figure 3: Ordered effect-size plot for the top-ranked (a) bettong activity model and (b) predator presence model. *day* – day since one-way gates closed in bettong activity model and day since gates first opened in the predator presence model. The proportion of models the variable is included in out of six AICc top-ranked models is bracketed next to each variable in a)

Figure 4: Fitted response plot for AIC-top-ranked model of predator presence, showing response of probability of predator presence to bettong tracks per day per transect in Spring 2017 (solid) and Autumn 2018 (dotted line) survey seasons.

Discussion

Reintroduction of threatened mammals beyond predator-free havens is important for their ongoing conservation (Short *et al.* 1992; Christensen & Burrows 1994). Yet, despite close to 1,600 bettongs exiting the one-way gates in our study, few remained at monitoring sites for longer than three months, with most disappearing within a month after gates were closed in either survey season. We located many bettong carcasses and signs of predation near one-way gates. After gates were closed, bettong track counts declined most rapidly when predators were present, indicating that mortality due to predators, rather than movement away from gate sites is likely to be responsible for the rapid decline in activity. Although bettongs did not persist for any substantial length of time on survey transects after gates were closed, we found that supplementary feeding, vegetation cover and shelter availability had a positive effect on bettong activity, as hypothesised, indicating that if predators could be reduced or removed, supplementary feeding would be a good approach to avoiding hyperdispersal and act as a form of soft release.

Supplementary feeding

Bettong track counts were higher at feeding sites compared with non-feeding sites, supporting the hypothesis that feeding encourages site fidelity which reinforces similar findings for other species such as prairie voles, dormice and cotton rats (Cole & Batzli 1978; Bright & Morris 1994; Doonan &

Slade 1995). However, counts declined more quickly at feeding sites, indicating feeding was not associated with increased residence time. Supplementary food may have enabled bettongs to feed more efficiently at night and so spend less time exposed to predators outside shelter and/or reduced competitive pressure to disperse. However, it did not ultimately result in substantially better outcomes for bettongs in the presence of high predator pressure.

While feeding had a positive effect on bettong site residence after gates were closed, the association between high numbers of predators and areas with high bettong activity was concerning and indicates a potential trade-off. While we did not explicitly test for it in our study, it is possible that predators target feeding stations frequented by prey species (e.g. Dunn and Tessaglia-Hymes, 1994). Future studies should seek to clarify whether feeders benefit prey persistence in the presence of predators and under varying levels of predator control. Supplementary feeding accompanied by targeted predator control around feeders may be an effective way to distribute predator control resources and increase bettong residence time.

Predation

Bettong track counts were lower in surveys when predators (cats, foxes or dingoes) were present at non-feeding sites but higher at feeding sites. Similarly, predators were more likely to be detected when bettong track counts were high. This suggests that predators were associated with high levels of bettong activity either because predators were attracted to the prey activity or because both bettongs and predators are attracted to similar habitat attributes (e.g. rabbit warrens for shelter) (Moseby *et al.* 2009). Feral cats and foxes are known to travel long distances to access plentiful food resources (Bubela *et al.* 1998; Tsukada 1997; McGregor *et al.* 2015). In our study, predator activity at sites was not measured prior to the gates being open so we cannot categorically state that predators were attracted to sites where bettong activity was high rather than being attracted to similar site attributes. However, it would appear that the strong relationship between predator presence and bettong track count indicates prey-tracking by predators is quite likely.

Evidence of predation at study sites (bettong carcasses) reinforces the primary role of predation in the rapid demise of bettongs, in line with previous reintroduction studies (Christensen & Burrows 1994; Short & Turner 2000; Moseby *et al.* 2011; Bannister *et al.*, 2016). We regularly found bettong carcasses and remains near the fenceline with tracks and puncture wounds indicating predation by wedge-tailed eagles, dingoes and cats. Most carcasses suspected to be killed by dingoes were largely intact, consistent with the known hunting behaviour of dingoes on bettongs and their tendency to surplus kill (Thomson 1992; Moseby *et al.* 2011). Predation by dingoes and wedge-tailed eagles appeared to be highest immediately after bettongs exit gates, when animals had not yet established or located burrows and were easier prey for predators that hunt using visual cues. Camera trap images of cats and dingoes investigating one-way gates suggest the predators were aware of gates. We did not observe any conclusive evidence of fox predation. Fox tracks were recorded less frequently than cat and dog tracks, potentially due to their avoidance of dingoes at a fine spatial scale (Mitchell & Bank 2005; Moseby *et al.* 2012).

More intensive and sustained predator control should be tested in the future to determine if bettongs can survive outside of the reserve when supportive measures such as feeding and supplementary shelter are combined with intensive predator control. Aerial baiting was not conducted during our study and despite ground baiting, shooting and trapping occurring around the perimeter of the reserve, predator control was relatively localised. Previous attempts to use aerial baiting to control predators around AR were unsuccessful at controlling cats but successfully controlled foxes (Moseby & Hill 2011). Intensive control could include aerial baiting for foxes, Felixer grooming traps and leghold traps for feral cats and rabbit control to naturally reduce predator abundance. These activities could occur within at least a 10 km radius of the reserve to reduce re-invasion rates. There is a precedent for bettongs co-existing with low densities of cats - 0.46 cats per square km within a large enclosed area (Moseby *et al.* 2018b). This density is higher than the estimated average density of feral cats across Australia (0.27 cats per square km) (Legge *et al.* 2017). The role of dingoes in reintroduction of critical-weight-range mammals also requires further research. While dingoes and bettongs co-existed prior to European settlement, it is unclear how to manage dingoes at open-landscape reintroduction sites given the evidence of dingo predation on bettongs from this study and others (Bannister *et al.* 2016). Improved understanding of co-existence thresholds for vulnerable native prey such as bettongs, and the influence of other factors such as the availability of alternative prey, will guide efforts for reintroductions outside of predator-free areas.

Dispersal

In addition to predation, dispersal may have been a factor in the relatively short residence time of bettong sign on transects. Bettongs are social animals, and some individuals may have moved away from gates due to hyperdispersal after being separated from their warren group (Kleiman 1989; Thompson *et al.* 2001) or gradually dispersed to dunes further from their exit point as more bettongs exited the gates and density increased (Moseby *et al.* 2011; West *et al.* 2018). The distance between dunes was well within the home range radius of this species (Finlayson & Moseby 2004). However, the positive correlation between bettong tracks and number of exits at nearby gates suggests that most animals were not hyperdispersing but staying close to the point at which they exited the reserve, at least initially.

Three times more bettongs dispersed through one-way gates in 2017 than in 2018, mainly due to gates being open for a longer period (six weeks in 2017 compared to two weeks on the West boundary or four weeks on the East boundary in 2018). Drier conditions likely influenced the behaviour of bettongs inside and outside of the reserve, the vegetation cover available for use and bettongs' reliance on supplementary food. The recent dry conditions also mean we will need to wait until significant rains to test the effect of exit of 1600 bettongs from the reserve on vegetation recovery within the reserve. Exit rates were slightly lower in 2018, likely because of previous exits by bettongs with warrens close to one-way gates. Consistently higher rates of exits through gates on the East boundary and higher local bettong activity were likely due to all gates on this side being newly opened (gates on the west side had been in use previously), and the presence of free water from a leaking pipeline outside the eastern fence. The effect of distance to free water is not explicitly modelled here. Free water only existed outside the eastern fence, so its effect would be statistically confounded by other differences between east and west sides of the reserve.

Habitat

As predicted, bettong activity was higher at sites with greater mid-storey and canopy vegetation cover. Mid-storey cover is likely to protect bettongs from predators and/or provide food. However, this effect was not statistically significant, potentially because there was low variation in shrub cover across sites and low cover generally. Canopy cover was a weak predictor for bettong tracks and is likely to be a less important resource for bettongs. Our hypothesis that bettong activity would be higher at sites with more rabbit warrens was supported, reinforcing that shelter is a vitally important resource for bettongs and the bettongs will use burrows constructed by rabbits (Robley et al. 2002).

Management implications and future research

Bettong track counts were higher and declined more quickly when food was provided. This suggests that a higher number of individuals were supported at feeding sites and these animals dispersed or were predated faster than at non-feeding sites. However, we were unable to directly test these hypotheses as individual bettongs were not radio-collared.

This study reinforces understanding that supplementary feeding can enhance activity (and potentially site fidelity) in reintroduced species but is unlikely to increase residence time in the absence of intensive predator control (Rickett *et al.* 2013; Bannister *et al.* 2016; De Milliano *et al.* 2016). Re-establishing self-sustaining populations of locally extinct species such as bettongs outside predator-free areas may be possible if habitat requirements are met and sufficient predator control is applied, however this remains challenging (Moseby *et al.* 2011). Future studies could use a stratified experimental design to identify the interactions between feeding, habitat and minimum predator thresholds. Quantifying predation rates close to and remote from feeders would assist in determining the net benefit of soft-release feeding. Importantly, differentiating between mortality and bettong dispersal away from one-way gate sites is essential and would require radio-collaring and close monitoring of bettongs that exited gates.

Management of overabundant populations of threatened species within predator-free havens remains a challenge. As yet, there is no evidence for density dependent reductions in reproductive rates for burrowing bettongs (Finlayson & Moseby 2004), and it is clear that such population regulation is insufficient to maintain the bettong population at a level that does not negatively impact vegetation condition and other threatened species (Moseby et al. 2018). One-way gates facilitating dispersal out of fenced reserves are effective at reducing the density of bettongs within the reserve (Arid Recovery, unpublished data) but the difficulty of establishing dispersing animals outside the fence must be considered in balancing conservation and animal welfare goals in managing confined populations. A more sustainable option currently being tested may be regulation by introduction of native mammalian predators (West et al. 2019).

References

Akaike, H 1973, ‘Information theory and an extension of the maximum likelihood principle’, in: Parzen, Tanabe K. & Kitagawa G (eds), *Selected Papers of Hirotuga* Akaike (Perspectives in Statistics), Springer Series, New York, 199-213

Algar D., Onus M. & Hamilton N. (2013). Feral cat control as part of Rangelands Restoration at Lorna Glen (Matuwa), Western Australia: the first seven years. *Conservation Science Western Australia* **8**, 367-381.Bannister H. L., Lynch C. E. & Moseby K. E. (2016) Predator swamping and supplementary feeding do not improve reintroduction success for a threatened Australian mammal, *Bettongia lesueur*. *Australian Mammalogy* **38**, 177-187.

Barton K. (2013). MuMIn: multi-model inference, rCRAN package

Bates D., Maechler M., Bolker B & Walker, S (2014) Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software* **67**, 1-48.

Bennett V. A., Doerr V. A. J., Doerr E. D., Manning A. D., Lindenmayer D. B., and Yoon H. (2013) Habitat Selection and behaviour of a reintroduced Passerine: Linking experimental restoration, behaviour and habitat ecology, *PLOS one* **8**, 1-12.

Bice J. & Moseby K. (2008) Diets of the re-introduced greater bilby (*Macrotis lagotis*) and burrowing bettong (*Bettongia lesueur*) in the Arid Recovery Reserve, Northern South Australia. *Australian Mammalogy* **30**, 1-12.

Blumstein D. T., Mari M., Daniel J. C., Ardron J G., Griffin A.S. & Evans C.S. (2002) Olfactory predator recognition: wallabies may have to learn to be wary. *Animal Conservation* **5**, 87-93.

Bright P. W. & Morris, P. A. (1994) Animal translocation for conservation: performance of dormice in relation to release methods, origin and season. *Journal of Applied ecology*, 699-708.

Bubela T., Bartell R. & Müller W. (1998) Factors affecting the trappability of red foxes in Kosciusko National Park. *Wildlife Research* **25**, 199-208.

Burbidge A. A. & McKenzie N. L. (1989) Patterns in the Modern Decline of Western Australia's Vertebrate Fauna: Causes and Conservation Implications. *Biological Conservation* **50**, 143-198.

Bureau of Meteorology 2019, Climate statistics for Australian locations: summary statistics Roxby Downs, viewed 7th July 2019, http://www.bom.gov.au/climate/averages/tables/cw_016096.shtml

Butler, K., Paton, D. and Moseby, K. (2019). One‐way gates successfully facilitate the movement of burrowing bettongs (*Bettongia lesueur*) through exclusion fences around reserve. *Austral Ecology*, **44**, 199-208.

Cavanaugh J. E. (1997) Unifying the derivations of the Akaike and corrected Akaike information criteria, *Statistics & Probability Letters*, **31**, 201–208.

Chapman T. F. (2015) Comparison of soils and plants on the active and relic parts of a recolonised burrowing bettong (*Bettongia lesueur*) warren. *Pacific Conservation Biology* **21**, 298-306.

Christensen P. & Burrows N 1994, 'Project desert dreaming: experimental reintroduction of mammals to the Gibson Desert, Western Australia', in: Serena M (eds), *Reintroduction Biology of Australian and New Zealand Fauna*, Surrey Beatty & Sons, Sydney, Australia, 199-207

Christensen P. E., Ward B. G. & Sims C. (2013) Predicting bait uptake by feral cats, *Felis catus,* in semi-arid environments, *Ecological Management and Restoration* **14**, 47-53.

Cole F. R. & Batzli G. O. (1978) Influence of supplemental feeding on a vole population. *Journal of Mammalogy* **59**, 809-819.

Crisp H. & Moseby K. (2010) One-way gates: Initial trial of a potential tool for preventing overpopulation within fenced reserves. *Ecological Management and Restoration* **11**, 139-141.

D'Antionio M., Hughes R. F. & Tunison J. T. (2011) Long-term impacts of invasive grasses and subsequent fire in seasonally dry Hawaiian woodlands, *Ecological Applications* **21**, 1617-1628.

Department of Environment 2019. *Bettongia lesueur lesueur* in Species Profile and Threats Database, viewed 23rd March 2019, http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=66657

Doonan T. J. & Slade, N. A. (1995) Effects of supplemental food on population dynamics of cotton rats, *Sigmodon hispidus*. *Ecology* **76**, 814-826.

Dunn E. H. & Tessaglia-Hymes D. L. (1994) Predation of birds at feeders in winter. *Journal of Field Ornithology* **65**, 8–16.

Finlayson G. R. & Moseby K. E. (2004) Managing confined populations: the influence of density on the home range and habitat use of reintroduced burrowing bettongs. *Wildlife Research* **31**, 457-463.

Hardman B., Moro D. & Calver M. (2016) Direct evidence implicates feral cat predation as the primary cause of failure of a mammal reintroduction programme. *Ecological Management and Restoration* **17**, 152-158.

Harrison X. A. (2014), Using observation-level random effects to model overdispersion in count data in ecology and evolution. *PeerJ* **2**, e616.

Hartig F 2018, *DHARMa: Residual diagnostics for hierarchical (multi-level/mixed) regression models version 0.2.0,* viewed 18th October 2018, http://florianhartig.github.io/DHARMa/

Hayward M. W., Moseby K. E. & Read J. L. (2014) 'The role of predator exclosures in the conservation of Australian fauna', in: Glen A. S & Dickman C. R (eds), *Carnivores of Australia: Past, Present and Future,* CSIROPublishing, Melbourne, Australia, 353-371

Hayward M. W., O'Brien J. & Kerley G. I. H. (2007) Carrying capacity of large African predators: predictions and tests, *Biological Conservation* **139**, 219-229.

IUCN 2012, *IUCN/SSC Guidelines for re-introductions and other Conservation Translocations*, IUCN/SSC Re-introduction Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK, viewed 23rd March 2019, < https://www.iucn.org/content/new-guidelines-conservation-translocations-published-iucn>.

James A. I. & Eldridge D. J. (2007) Reintroduction of fossorial native mammals and potential impacts on ecosystem processes in an Australian desert landscape. *Biological Conservation* **138**, 351-359.

James A. I., Edridge D. J. & Hill B. M. (2009). Foraging animals create fertile patches in an Australian desert shrubland, *Ecography* **32**, 723-732.

Kanowski J., Roshier D., Smith M., Fleming. A. (2018). Effective conservation of critical weight range mammals: reintroduction projects of the Australian Wildlife Conservancy. In: *Recovery in Australian Threatened Species, Book of Hope* ch 28, CSIRO Publishing

Kleiman D. G. (1989) Reintroduction of captive mammals for conservation. Guidelines for introducing endangered species into the wild. *Bioscience* **39**, 152– 161.

Legge S., Murphy B. P., McGregor H., Woinarski J.C.Z., Augusteyn J., Ballard G., Baseler M., Buckmaster T., Dickman C.R., Doherty T. and Edwards, G. (2017) Enumerating a continental-scale threat: how many feral cats are in Australia? *Biological Conservation* **206**, .293-303.

Legge S., Woinarski J., Burbidge A., Palmer, R Ringma., J, Radford J., Mitchell N., Bode M., Wintle B., Baseler M., Bentley J., Copley P., Dexter N., Dickman C., Gillespie G., Hill B., Johnson C., Latch

P., Letnic M., Manning A., McCreless E., Menkhorst P., Morris K., Moseby K., Page M., Pannell P. & Tuft K. (2018) Havens for threatened Australian mammals: the contributions of fenced areas and offshore islands to the protection of mammal species susceptible to introduced predators. *Wildlife Research* **45**, 627-644

Linley G. D., Moseby K.E. & Paton D. C. (2017) Vegetation damage caused by high densities of burrowing bettongs (*Bettongia lesueur*) at Arid Recovery. *Australian Mammalology* **39**, 33–41.

McGregor H. W., Legge S., Potts J., Jones M. E. & Johnson, C. N. (2015) Density and home range of feral cats in north-western Australia. *Wildlife Research **42***, 223-231.

De Milliano J., Di Stefano J., Courtney P., Temple-Smith P., & Coulson G. (2016). Soft-release versus hard-release for reintroduction of an endangered species: an experimental comparison using eastern barred bandicoots (*Perameles gunnii*). *Wildlife Research* **43**, 1-12.

Mitchell A. M., Wellicome T. I., Brodie D. & Cheng K. M. (2011) Captive-reared burrowing owls show higher site-affinity, survival and reproductive performance when reintroduced using a soft-release. *Biological Conservation* **144**, 1382-1391.

Mitchell B. D. & Banks P. B. (2005) Do wild dogs exclude foxes? Evidence for competition from dietary and spatial overlaps. *Austral Ecology* **30**, 581–591.

Moseby K. E. & Hill B. M. (2011) The use of poison baits to control feral cats and red foxes in arid South Australia 1. Aerial baiting trials. *Wildlife Research* **38**, 338–349.

Moseby K. E., Hill B. M. & Lavery T. H. (2014) Tailoring release protocols to individual species and sites: one size does not fit all. *PLoS One* **9**, e99753.

Moseby K. E., Lollback G. W. & Lynch, C. E. (2018a) Too much of a good thing; successful reintroduction leads to overpopulation in a threatened mammal. *Biological Conservation* **219**, 78-88.

Moseby K. E., Letnic M., Blumstein D.T. & West, R. (2018b) Understanding predator densities for successful co-existence of alien predators and threatened prey. *Austral Ecology* **44**, 409-419Moseby K. E., Neilly H., Read J. & Crisp H. (2012) Interactions between a Top Order Predator and Exotic Mesopredators in the Australian Rangelands. *International Journal of Ecology*, 1-15

Moseby K. E. & Read J. L. (2006). The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biological Conservation* **127**, 429-437.

Moseby K. E., Read J. L., Paton D. C., Copley P., Hill B. M. & Crisp. H. A. (2011a) Predation determines the outcome of 10 reintroduction attempts in arid South Australia. *Biological Conservation* **144**, 2863-2872.

Moseby K. E., Stott J. & Crisp H. (2009). Improving the effectiveness of poison baiting for the feral cat and European fox in northern South Australia: The influence of movement, habitat use and activity. *Wildlife Research* **36**, 1-14.

Noble J. C., Muller. W. J., Detling. J. K. & Pfitzner. G. H. (2007) Landscape ecology of the burrowing bettong: Warren distribution and patch dynamics in semiarid eastern Australia. *Austral Ecology* **32**, 326-337.

Ostermann S. D., Deforge J. R. & Edge W. D. (2001) Captive breeding and reintroduction evaluation criteria: a case study of Peninsula Bighorn Sheep, *Conservation Biology* **15**, 749-760.

R Core Team 2013, *R: a language and environment for statistical computing*, viewed 11th October 2018, http://www.R-project.org/

Rickett J., Dey C. J., Stothart J., O'Conner C. M., Quinn J. S. & Weihong J. (2013) The influence of supplementary feeding on survival, dispersal and competition in translocated brown teal, or pateke (*Anas chlorotis*). *Emu* **113**, 62 –68.

Robley, A., Short, J., and Bradley, J.S. (2002) Do European rabbits (*Oryctolagus cuniculus*) influence the population ecology of the burrowing bettong (*Bettongia lesueur*)? *Wildlife Research* **29**, 423-429.

Short J., Bradshaw S. D., Giles J., Prince R. I. T. & Wilson G.R. (1992) Reintroduction of macropods (Marsupialia: Macropodoidea) in Australia, a review. *Biological Conservation* **62**, 189-204.

Short J. & Smith A. (1994). Mammal Decline and Recovery in Australia. *Journal of Mammalogy* **75**, 288-297.

Short J. & Turner B. (2000) Reintroduction of the burrowing bettong *Bettongia lesueur* (Marsupialia: Potoroidae) to mainland Australia. *Biological Conservation* **96**, 185-196.

Slotow R., Garai M. E., Reilly B., Page B. & Carr R. D. (2005) Population dynamics of elephants re-introduced to small fenced reserves in South Africa. *South African Journal of Wildlife Research* **35**, 23-32.

Thompson J. R., Bleich V. C., Torres S. G. & Mulcahy G. P. (2001) Translocation techniques for mountain sheep: does the method matter? *The South West Naturalist* **46**, 87–93.

Thomson P. C. (1992) The behavioural ecology of dingoes in north-western Australia. III. Hunting and Feeding behaviour, and diet. *Wildlife Research* **19**, 531-541.

Tsukada H. (1997) A division between foraging range and territory related to food distribution in the red fox. *Journal of Ethology* **15**, 27.

Tuberville T., Clark E., Buhlmann K. & Gibbons J. (2005) Translocation as a conservation tool: site fidelity and movement of repatriated gopher tortoises (*Gopherus Polyphemus*). *Animal Conservation* **8**, 349-358.

Waller N. L., Gynther I., Freeman A. B., Lavery T. H. & Leung L. K.(2017) The Bramble Cay melomys *Melomys rubicola* (Rodentia: Muridae): a first mammalian extinction caused by human-induced climate change? *Wildlife Research* **44**, 9-21.

West R., Letnic M., Blumstein D. & Moseby. K. E. (2018) Predator exposure improves anti- predator responses in a threatened mammal. *Journal of Applied Ecology* **55**, 147-156

West, R. S., Tilley, L., & Moseby, K. E. (2019). A trial reintroduction of the western quoll to a fenced conservation reserve: implications of returning native predators. *Australian Mammalogy*

Woinarski J. C. Z., Burbidge A. A. & Harrison P. L. (2015) Ongoing unravelling of a continental fauna: decline and extinction of Australian mammals since European settlement. *Proceedings of the National Academy of Sciences of the United States of America* **112**, 4531-4540.

Tables

Table 1: Summary of independent variables offered to the models. '*' indicates the reference level for categorical variables

Predictor	Predictor type	Description

Bettong activity		Sum of tracks on a transect for a given day
day since gates closed	Continuous	
survey season	Factor	Spring 2017*/Autumn 2018
one-way gate exits	Continuous	cumulative total exits
feeding treatment	Factor	non-feeding*/feeding treatment
midstorey vegetation cover	Factor	*low, high,* relative to median cover value for this height class (10.5%)
canopy vegetation cover	Factor	*low, high,* relative to median cover value for this height class (2.75%)
shelter availability	Continuous	number of warrens at a site
predator presence	Factor	daily absence*/presence of cat, fox and/or dingo tracks on survey transect
site and East/West boundary	Categorical	random intercept effect to account for the nestedness of the survey design (sites grouped along East and West boundaries, with repeat surveys of each site within each survey season)
Predator occurrence		
bettong track count	Continuous	
survey season	Factor	Spring 2017*/Autumn 2018
treatment	Categorical	non-feeding*/feeding treatment
day since gates opened	Continuous	
site within East/West boundary	Categorical	random intercept effect to account for the nestedness of the survey design (sites grouped along East and West boundaries, with repeat surveys of each site within each survey season)

Table 2: Top-ranked models (ΔAICc ≤ 2) of bettong activity and predator occurrence, relative to the null model. Predictors are described in Table 1. 'Top-ranked' referes to the bettong activity model including all variables in the top six dredged models. '*' indicates a statistically significant relationship (< 0.05). df refers to model degrees of freedom; weight to the probability that a given model is the best, relative to other models; R^2m the marginal R-squared value (conditional on fixed variables); R^2c the conditional R-squared value (conditional on fixed and random variables)

Bettong Activity

	predator presence	*shelter*	*total exits*	*food*	*canopy vegetation*	*shrub cover*	*day:predator presence*	*day:food*	*predator presence:food*	*df*	*Δ AICc*	*weight*	*R²m*	*R²c*
top-ranked	-0.69*	0.73*	0.13	0.84*	0.01	0.23	0.72	-0.45	0.59*	17			0.91	1.00
1	-0.63*	0.89*		0.51*		0.16*			0.77*	13	0.00	0.12	0.90	1.00

2	-0.63*	0.88*		0.47*		0.16*		0.19	0.77*	14	0.60	0.09	0.90	1.00
3	-0.64*	0.88*		0.51*	0.03	0.16*			0.78*	14	1.08	0.07	0.90	1.00
4	-0.66*	0.88*	-0.04	0.50*		0.15*			0.78*	14	1.66	0.05	0.90	1.00
5	-0.64*	0.88*		0.46*	0.03	0.16*		0.19	0.78*	15	1.69	0.05	0.90	1.00
6	-0.61*	0.89*		0.51*		0.16*	-0.10		0.77*	14	1.75	0.05	0.90	1.00
null										5	5353.87	0.00	0.81	1.00

Predator Occurrence

	bettong track count	*day since gates opened*	*food*	*season*	*df*	*Δ AICc*	*weight*	*R^2m*	*R^2c*
1	0*	0.04*		1.7*	6	0	0.52	0.17	0.35
2	0.01	0.04*	-0.26	1.76*	7	1.56	0.24	0.17	0.36
null					3	50.23	0	0	0.11

Table 3: Proportion of survey days predators were recorded on transects in Spring 2017 and Autumn 2018 and averaged across both seasons at Arid Recovery Reserve, South Australia.

Survey	all predators	cats	foxes	dingoes	N surveys
2017	0.67	0.63	0.10	0.10	144
2018	0.81	0.76	0.02	0.16	99
Average	0.74	0.70	0.06	0.13	