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Targeted sampling successfully detects the cryptic and declining arboreal marsupial *Phascogale pirata* in northern Australia

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Abstract

The threatened northern brush-tailed phascogale (*Phascogale pirata*) is one of the most poorly known mammals in Australia. While the few available records indicate a decline in its distribution and abundance, it has not previously been subject to intensive targeted survey. Here, we trialled a specifically-tailored methodology for detection of *P. pirata*, with the aim of informing ongoing survey and monitoring of this species. We deployed 50 motion-sensor cameras (spaced closely together in a grid 500 × 1000 m) on Melville Island (Northern Territory, Australia), between June 2018 and May 2019. Cameras were baited and secured to trees ~3 m above the ground on a bracket facing the trunk. We selected for large (>30 cm diameter at breast height [DBH]) trunks of the dominant tree species (*Eucalyptus miniata*, *E. tetrodonta* and *Corymbia nesophila*). We detected *P. pirata* 16 times on eight cameras over the duration of the study, finding that detection was most likely on large (DBH >41.5 cm) *E. tetrodonta* trees during the wet season. Our results indicate that survey effort for this species should be seasonally targeted and focused on large trees. However, the efficacy of additional methods (nest boxes, Elliott traps) and a comparison between detections on arboreal versus ground-based cameras requires further investigation. We highlight the importance of conducting additional work on this species, as little is known about its ecology, population trends and threats, making it difficult to assess its conservation status. Without more targeted work, *P. pirata* is at risk of slipping into extinction unnoticed.

Additional keywords northern brush-tailed phascogale, targeting monitoring, rare and cryptic species, motion-sensor cameras, Tiwi Islands, tree-traps.

Summary text

The northern brush-tailed phascogale (*Phascogale pirata)* has likely declined across its entire mainland range (monsoonal Northern Territory), but this is difficult to quantify due to the very limited baseline data available. We conducted the first targeted survey for *P. pirata* on Melville Island in the Northern Territory, finding that detection was more likely on large *Eucalyptus tetrodonta* trees in wetter months (i.e. October–April). Our approach (using motion-sensor cameras mounted on tree trunks) may be used to determine whether populations have persisted across the historic range or (where guided by suitable habitat mapping) to search for new populations.

Running head targeted survey detects *Phascogale pirata*

Introduction

Effective biodiversity conservation depends on adequate information regarding species' distributions and population trends, which is sometimes difficult to obtain. This can be particularly true for threatened species, which often occur at low densities or at few locations (Legge *et al.* 2018). Broad-scale general wildlife surveys rarely produce the data required to infer population trends for all of the species on which they report. This is in part due to imperfect detection, and because some species are cryptic by nature (Einoder *et al.* 2018). In such cases, general wildlife surveys can preclude the early detection of decline, which severely limits our capability to implement effective and timely remedial management (Claridge *et al.* 2005). It is therefore critical that survey and monitoring of cryptic, threatened species is sufficiently intensive and targeted, with the design and methods tailored to suit the specific attributes of the target species (Legge *et al.* 2018).

The northern brush-tailed phascogale (*Phascogale pirata*) is one of Australia's most elusive and poorly known mammals. In part, this may be because it has only recently been (re-) recognised as a species distinct from the far more widespread and abundant *P. tapoatafa* (Aplin *et al*. 2015). It is endemic to the coastal, mesic savannas of the Northern Territory, a region known as the ‘Top End’. The limited available evidence indicates a decline in both distribution and population size (Woinarski *et al.* 2008; Woinarski *et al.* 2014), including the probable extirpation of one of its two known island populations (Woinarski *et al*. 2011), and a likely marked reduction in abundance in the largest conservation reserve from which it was known, Kakadu National Park (Woinarski *et al.* 2001; 2010). In the last decade it has been recorded from very few (<5) locations (Fig. 1). While an ongoing decline seems likely, extensive fauna surveys undertaken across the Top End (Woinarski *et al.* 2001; 2003; 2004; 2010; Firth *et al*. 2006; Davies *et al.* 2017) have produced only a handful of records, making it very difficult to quantify population trends.

Prior to this study, many of the most recent capture records of *P. pirata* were from the 5788 km^2 Melville Island in Australia's Northern Territory. Unlike parts of the species' mainland distribution, the Tiwi Islands (Melville Island and the adjacent Bathurst Island) still contain their complete pre-European mammal assemblage, with many threatened species remaining more abundant there than in their mainland range (e.g. Firth *et al.* 2006; Davies *et al.* 2017, Davies *et al.* 2018a). Despite this, an extensive general wildlife survey of 351 sites (236 on Melville Island and 115 on Bathurst Island) undertaken between 2000 and 2002 captured only two individuals of *P. pirata* in live-traps over 25 272 trap-nights (16 056 on Melville and 8280 on Bathurst), and reported none in spotlight searches of those sites (Firth *et al.* 2006). Both captures were on Melville Island, and there have been no confirmed reports of *P. pirata* on Bathurst Island (Aplin *et al.* 2015). More recently in 2015, a survey sampling 88 sites on Melville Island (86 of which had been previously sampled in 2000–2002), detected only one individual (in a cage trap) over almost 23 500 trap nights using live-traps and camera-traps (H. Davies, unpublished data; see Davies *et al.* 2018a for details on methodology). A concurrent study aimed at testing the importance of pyrodiversity for native mammals in 18 experimental fire plots across Melville Island (Davies *et al.* 2018b) was more successful, capturing a total of nine independent detections (H. Davies, unpublished data). That study deployed four camera-traps in each of three experimental fire treatments (burnt annually, burnt triennially and unburnt), replicating each treatment six times. All nine detections of *P. pirata* were from three nearby plots at a single site (located at Taracumbi, Fig. 1), and unlike previous sampling conducted on Melville Island, cameras were deployed for an extensive period (approximately two years, for a total effort of 8205 trap nights across the three Taracumbi plots and 48 627 trap nights across all sampled sites).

It is possible that conventional methods are not suitable for reliably detecting *P. pirata*, and this is supported by several studies conducted on a closely related species. In south-eastern Australia, large aluminium Elliott traps (48 × 15 × 16 cm), fixed to wooden T-shaped brackets and nailed to trees (large rough-barked eucalypts) approximately 2 m from the ground, are used for long-term monitoring of *P. tapoatafa* (Holland *et al.* 2012). This design has been shown to produce sufficient data from which population trajectories may be rigorously quantified. In Manjimup, 300 km south of Perth, Western Australia, the most useful method for detecting *P. tapoatafa* was found to be the establishment and ongoing inspection of nest boxes placed at a height of 3–4 m (Rhind and Bradley 2002). Scida and Gration (2017) trialled three different methods for sampling and monitoring *P. tapoatafa*; (i) live-traps fixed to trees approximately 2 m off the ground; (ii) remote cameras mounted 2–3 m away, facing hollow bearing trees or trees with loose or fibrous bark; and (iii) nest boxes placed 4–8 m above the ground. All three methods successfully detected *P. tapoatafa*, with remote cameras yielding the highest detection probabilities.

Given the general lack of historical records of *P. pirata*, it is likely that specifically tailored sampling methods are required to produce accurate and informative data. There has been only limited previous trialling of targeted methods for *P. pirata*. Ward (2006) installed 40 nest boxes at four sites to try to detect the species on West Island (Sir Edward Pellew group), but no individuals were recorded, possibly because it was already extirpated from the island. In the Top End’s Garig Gunak Barlu National Park, *P. pirata* was not detected in any of the 200 nest boxes installed, nor was it detected on images from the 75 nest box-associated cameras deployed from 2016–2018 (a total of 33 392 trap nights, L.A.Woolley, unpublished data). In Kakadu National Park (as part of the Kakadu Fauna Survey, Braithwaite 1985), six of the 20 Elliott traps deployed at each of 30 sites were set in trees (~2 m from the ground, taped to long nails hammered into trunks) to survey for arboreal mammals. However, this was found to be ineffective (as it did not increase the known species richness across sites), and no individuals of *P. pirata* were captured using this approach. Here, we investigate the capability of an explicitly-designed sampling protocol to detect *P. pirata,* and use the results to make recommendations for ongoing survey and monitoring of this threatened species.

Materials and methods

Study location

Melville Island is the larger of the two main Tiwi Islands, located approximately 80 km north of Darwin, in the Northern Territory, Australia. The island is of low relief (≤ 103 m above sea level) and experiences a highly seasonal (wet–dry tropical monsoonal) climate. Total annual rainfall varies substantially across the island, but at our site (see below) average yearly rainfall is approximately 1750 mm, of which > 95% is recorded between October and April (Bureau of Meteorology 2019). Vegetation includes savanna woodland and open forest dominated by eucalypts (particularly *Eucalyptus miniata*, *E. tetrodonta*, and *Corymbia nesophila*), with a predominantly grassy understory. Shrub density is highly variable, controlled in part by the frequency and severity of fires (Russell-Smith *et al.* 2003; Woinarski *et al.* 2004).

Study species

Phascogale pirata is a carnivorous marsupial, with body length of approximately 15–27 cm and weight of approximately 150–200 g, with grey dorsal fur and distinctive long dark hairs on the tail (Woinarski *et al.* 2014) (Fig. 2). It is currently listed as Vulnerable on the IUCN Red List (Woinarski *et al.* 2019), and under relevant Australian legislation (i.e. the *Environmental Protection and Biodiversity Conservation Act 1999*). There have been no ecological or behavioural studies of *P. pirata*. However, based on the behaviour and ecology of the closely related *P. tapoatafa*, *P. pirata* is likely to be solitary, arboreal, and strictly dependent on woodlands for food (feeding primarily on large invertebrates found on or beneath loose fibrous bark) and tree hollows for shelter and nesting (Scarff *et al.* 1998; Holland *et al.* 2012).*Phascogale tapoatafa* typically occupies a large home range (Soderquist 1995), but this is likely dependent on the quality of habitat. A study by Van Der Ree *et al.* (2001) in the Northern Plains of Victoria found that the average home range of females was an eighth the size of that at other Victorian sites (ca. 5.0 ha), likely due to high productivity and high density of suitable hollow-bearing trees at this location. Based on information about the life history of *P. tapoatafa*, males of *P. pirata* are presumed to have an annual lifecycle, mating only for a single, short season (1–2 weeks), after which all males in the population die (Woinarski *et al.* 2014). Females may survive to breed in a second year (Woinarski *et al.* 2014), but generally experience elevated mortality rates while raising young as a result of unusually high maternal investment (compared to other dasyurids) (Soderquist 1993). Breeding takes place around June or July in *P. tapoatafa*, with little seasonal variation throughout its range (i.e. populations persisting in areas of the tropics, subtropics, and temperate regions of Australia breed around this time of year, Soderquist 1993). Almost all records of *P. pirata* are from lowland eucalypt forest and woodland dominated by *E. miniata* and *E. tetrodonta* (Woinarski *et al.* 2014).

Data collection

We conducted a targeted survey at Taracumbi (Fig. 1) from the 30th of June 2018 to the 4th of May 2019. This site was chosen due to having recent confirmed detections of *P. pirata* (from a camera-trapping survey conducted by H. Davies in 2013–2015). We established a 500 × 1000 m grid, consisting of 50 Reconyx HP2X Hyperfire 2 Professional Covert Infrared remote-sensor cameras (with 0.2 second trigger speed), spaced 50–200 m apart. Cameras were chosen because they are an efficient, non-invasive survey tool, requiring minimal input of labour (Welbourne *et al.* 2015) and are easily deployed over long periods of time, leading to higher probabilities of detection (DeBondi *et al.* 2011). Cameras were fixed to steel brackets (~60 cm long) using socket cap screws and secured to trees 3 m above ground so that the camera focal point was facing the trunk of the tree (i.e. 60 cm between the cameras and the trunk). Cameras were sited facing baits composed of cotton wadding soaked in a mixture of honey, peanut butter and linseed oil, which was then placed inside PVC pipe canisters, sealed on either end by a vent cowl (which was made up of a heavy-duty mesh material, allowing the scent to escape the otherwise sealed cannister), and secured to the bracket in the centre of the camera's focal point. Additionally, a mixture of honey and water was sprayed along the trunk of the tree from 2 m above the cameras to ground-level as a further attractant (as in van der Ree *et al.* 2001; Holland *et al.* 2012; and Scida and Gration 2017). Camera traps were baited twice over the duration of the study. Given the importance of tree hollows for nesting and breeding, and thus a greater likelihood of animal use of these trees, larger (> 30 cm diameter at breast height [DBH], 1.3 m) trees were opportunistically selected for camera trap placement, as Woolley *et al.* (2018) reported that more than 70% of trees occurring in the tropical savannas of the Top End with DBH > 30 cm have hollows. The dominant rough-barked species were targeted: *E. miniata* (although the rough bark is restricted to the lower trunk for this species), *E. tetrodonta* and *C. nesophila.* Cameras were programmed to trigger at any time of day, set to take five images per trigger with a one second delay, and re-armed immediately after each trigger.

Data analysis

A detection event was defined as a set of camera images separated by 30 minutes. We summarised the camera images as binary detection/non-detection histories, pooling the data by month so that there were 10 observations for each camera. We then used generalised linear models (GLMs) with binomial error structure and logit link to investigate the influence of four variables (tree species, tree DBH, average monthly rainfall, time since bait deployment; see Table 1 for details) on the probability of detecting *P. pirata*, where all possible variable combinations were assessed. We included rainfall as a predictor because it likely affects net primary productivity, which may impact the species richness and abundance of the canopy arthropods consumed by *P. pirata* (Noske and Franklin 1999). Continuous variables (DBH, average monthly rainfall and time since bait deployment) were centred and standardised by deducting the mean and then dividing by two times the standard deviation (Gelman 2008). We did not consider interactions due to the limited size of our dataset. Given the small sample size, model selection was based on a robust form of Akaike's Information Criterion, AIC_c, an index that favours both model fit and model simplicity (Burnham and Anderson 2002). Models with lower values of AIC_c are considered to have greater support relative to other models in the candidate set. The saturated model (i.e. including all explanatory variables) was evaluated by inspecting Pearson residuals plotted against fitted values to test for homogeneity, independence and model fit. Zero-inflated models were unnecessary because 10 000 simulations of the best model predicted a similar percentage of zeros to that of the observed dataset (Zuur *et al.* 2012).

We had reason to suspect that detectability of *P. pirata* might differ between the wet (summer) and dry (winter) seasons, based on previous records. To test this, we allocated each month to "wet" and "dry" categories, based on their average rainfall (Bureau of Meteorology 2019, Table 1). Wet season months were defined as those with average rainfall of >100 mm, while dry season months had an average monthly rainfall < 25 mm. We compared wet and dry season observations using a Wilcoxon matched-paired test. All analyses were conducted using R version 3.6.1 (R Core Team 2019).

Results

Fifteen cameras failed during the 10-month deployment, mostly due to leaks in the water-proof cases, leading to water damage. One camera was struck by lightning, and the lens of another was destroyed by a black-footed tree-rat (*Mesembriomys gouldii*). On average, the total length of deployment (when the remaining 35 cameras were collecting data) was 299 days ± 23.9 (SE), ranging from 209–307 days. There were several thousand detections of non-target species, including other arboreal and semi-arboreal mammals (e.g. common brushtail possum [*Trichosurus vulpecula*], sugar glider [*Petaurus breviceps*], black-footed tree-rat), birds (e.g. Torresian crow [*Corvus orru*], blue-winged kookaburra [*Dacelo leachii*], owlet nightjar [*Aegotheles cristatus*]), reptiles (e.g. green tree snake [*Dendrelaphis punctulata*], goannas [*Varanus* spp.]), and invertebrates [particularly ants]). *Phascogale pirata* was detected 16 times on eight cameras over the duration of the survey (a total of 10 455 trap nights) (Fig. 1).

We detected *P. pirata* on all three target tree species (but most frequently on *E. tetrodonta*), with an average DBH of 53.5 ± 10 cm (SE), ranging from 41.5–72.3 cm. All detections occurred between October and April (corresponding to what would typically be referred to as the 'build up' [October–November] and 'wet' seasons [December–April]).

No single model was clearly superior at explaining *P. pirata* detections. Tree species, DBH and average monthly rainfall appeared in the top three ranked models at least twice, with tree species appearing in all three models (Table 2). By contrast, time since bait deployment was absent from all three top ranked models (Table 2). Visualisation of the overall direction and effect sizes of the estimated relationships (regression coefficients) revealed very little difference among the candidate set of models, with the exception of time since bait deployment, which had both negative and positive regression coefficients (available as Supplementary Material, see S1). Given that there was no evidence to suggest a significant effect of time since bait deployment on the probability of detecting *P. pirata* (i.e. in every model where it appeared, confidence intervals overlapped zero), we focus here on the top ranked model (containing tree species, DBH and average monthly rainfall) for further inference. *Phascogale pirata* was more likely to be detected on *E. tetrodonta* trees than on *E. miniata* trees (p = 0.01, Fig. 3a), but there was no evidence to suggest a difference in *P. pirata* detection on *C. nesophila* and *E. miniata* trees (p = 0.77, Fig. 3a). There was limited evidence of a positive effect of DBH and average monthly rainfall on *P. pirata* detections, with detections more likely to occur on trees with larger diameter (p = 0.09, Fig. 3b) and in wetter months (p = 0.07, Fig. 3c).

There was a relationship between season (dry vs. wet) and *P. pirata* detections, with detection being far more likely in the wet season (p = 0.007, Fig. 4).

Discussion

General wildlife surveys serve an important purpose in ecology and conservation management by inventorying the flora and fauna of a particular region, or by capturing baseline data for experimental studies (Margules and Austin 1990; Legge *et al.* 2018). However, they may provide little information on many species that require specialised sampling techniques, are particularly shy or cryptic, have small population sizes, or are of conservation priority (all of which are characteristics evident for *P. pirata*). In such cases, a more fit-for-purpose approach to surveys and monitoring (where the sampling methodology and timing is targeted optimally to detect the species of interest, Woinarski 2018), may be required to gain important information about distribution and habitat requirements, as well as population dynamics and trends. Here we show that using a sampling protocol designed to target *P. pirata* could produce more precise data, while providing insight to inform the design of future survey and monitoring programs targeting this species.

We found that *P. pirata* was more likely to be detected on large (DBH > 41.5 cm), *E. tetrodonta* trees. This result was expected, given that the species is likely dependent on tree hollows for nesting and shelter (Woinarski *et al.* 2014), and hollows are more abundant in this environment in trees with DBH > 30 cm (Woolley *et al.* 2018). Additionally, *P. pirata* is thought to behave similarly to *P. tapoatafa,* which has been estimated to spend 80–90% of its time foraging in trees on or beneath the bark of trunks and major branches, coming to the ground only occasionally to prey on litter invertebrates (Traill and Coates 1993; Scarff *et al.* 1998). The rough, stringy bark of *E. tetrodonta* (which extends from the base of the tree to the canopy) may provide good foraging habitat, as well as good purchase for climbing.

We did not detect *P. pirata* between July and October (dry season), and the modelling identified a strong effect of season on the probability of detection (i.e. *P. pirata* was far more likely to be detected in the wet season). This could be a result of the semelparous breeding strategy of *P. pirata.* Like *P. tapoatafa,* there is some evidence to suggest that *P. pirata* breeds around June or July (Soderquist 1993). A recent capture (September 2019) of a lactating female in an Elliott trap on Melville Island (with no pouch young) provides some further support to this suggested seasonality of breeding (H. Davies, unpublished data). This is based on a period of ~30 days between insemination and birth and ~48 days before young are left in the nest while the mother forages (Soderquist 1993). It is therefore likely that the population was reduced (through near absence of males) around the onset of our camera deployment (early July), and that the remaining population was nesting, thus leading to a reduction in detectability. The increase in movement determined by the need for tree hollows by nesting females, as well as dispersal of young post-weaning, may also explain increased detectability in the wet season.

Breeding seasonality may help explain why historic records of *P. pirata* are so sparse. Most of the baseline survey work conducted in the Top End has been done during the dry season, primarily due to logistical challenges associated with access to field sites, as well as for safety reasons. Remote cameras may be useful for monitoring *P. pirata* at times of year when access is limited, however this too comes with challenges; over the duration of this study, we lost data from a total of 15 cameras (many of which were due to water damage). While this may be avoidable (by using additional water-proofing equipment), it is unclear how such modifications might affect different components of the camera (e.g. its ability to trigger), or animal responses, and this should be investigated. Furthermore, some things simply cannot feasibly be avoided (e.g. lightning strikes, cyclone damage), and thus there is an increased risk associated with conducting monitoring over the wet season which needs to be taken into consideration.

Targeted sampling, and in particular the use of camera traps, has led to some recent increase in the rate of detection of *P. pirata*. However, this should not be interpreted to indicate any trend of increase in abundance or distribution. Almost all recent detections of this species are from a small set of sites on Melville Island. Notably, *P. pirata* has not been recorded from the

Cobourg Peninsula (Garig Gunak Barlu National Park) since 2007 (Atlas of Living Australia 2018), notwithstanding the proximity of this area to Melville Island (~27 km), its similar cover of tall open forest, and substantial survey and monitoring effort. This effort has included a long-term nest box and camera monitoring program, which recently detected another threatened dasyurid (the fawn antechinus [*Antechinus bellus*]), during the wet season.

The causes of the decline in the distribution of *P. pirata* are not well resolved, but given the general decline in native mammals in the Northern Territory, it is likely that the species is affected by similar threats. These include habitat clearance (although this does not explain its decline in large conservation reserves such as Kakadu National Park), including on Melville Island for plantations of exotic timber species, changed fire regimes, poisoning by invasive cane toads (*Rhinella marina*), predation by feral cats (*Felis catus*), and habitat degradation caused by livestock and feral stock (Woinarski *et al.* 2011). The spatial patterning of decline of *P. pirata* in the monsoonal Northern Territory (Fig. 1) is comparable to the decline of several other native mammal species, with contraction to the high rainfall (northern) areas, and/or persistence on some islands (Ziembicki *et al.* 2015). For *P. pirata*, this may be because higher rainfall areas provide the most suitable habitat—they are more productive and support larger trees with more hollows (Woolley *et al.* 2018).

Recommendations for future survey and monitoring

Given the very few detections of *P. pirata*obtained during previous surveys conducted in the Top End (despite extensive survey effort), it is likely that targeted surveys are required for obtaining insight into the conservation status and ecology of this species. Further work is required to determine an optimal sampling regime that is capable of providing the necessary information to help guide the effective conservation of *P. pirata*. Notably, it is worthwhile testing the efficacy of ground-based motion-sensor cameras, set to face the trunks of trees (a technique yielding relatively high detection probabilities for *P. tapoatafa* in south-east Australia, Scida and Gration 2017). If appropriate, this approach would be simpler to implement logistically. Nevertheless, the technique described here would work to assess whether previously reported populations of this species have persisted and, where guided by mapping of habitat suitability, help survey for currently unknown populations. It would also provide a robust mechanism for monitoring responses to perceived threats (such as fire or cyclones) and management interventions. The sampling technique could be further complemented by use of nest boxes or tree-sited Elliott traps in order to provide additional information on population size and structure, breeding and genetics, disease and health status—parameters that are difficult or impossible to obtain from cameras alone.

Additional methods should be tested, and their efficacy assessed (both in terms of reliability and cost-effectiveness). However, regardless of the method used, surveys that aim to monitor *P. pirata* should include intensive effort, and target large, hollow-bearing, rough barked trees (notably *E. tetrodonta)* during the late dry and wet seasons.

General wildlife surveys aimed at targeting multiple species are unlikely to produce the data required to estimate population size, or to inform ongoing management of *P. pirata* and its habitat. At present, very little is known about the species' ecology, population trends, threats, and conservation status. More survey and monitoring is urgently needed, as the current lack of information constrains the direction and effectiveness of management responses. This study shows that with suitable, intensive and targeted sampling, such survey and monitoring is possible. Without an increase in targeted effort, this species could slip unnoticed into extinction.

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Conflicts of interest

The authors declare no conflicts of interest.

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Figures and tables

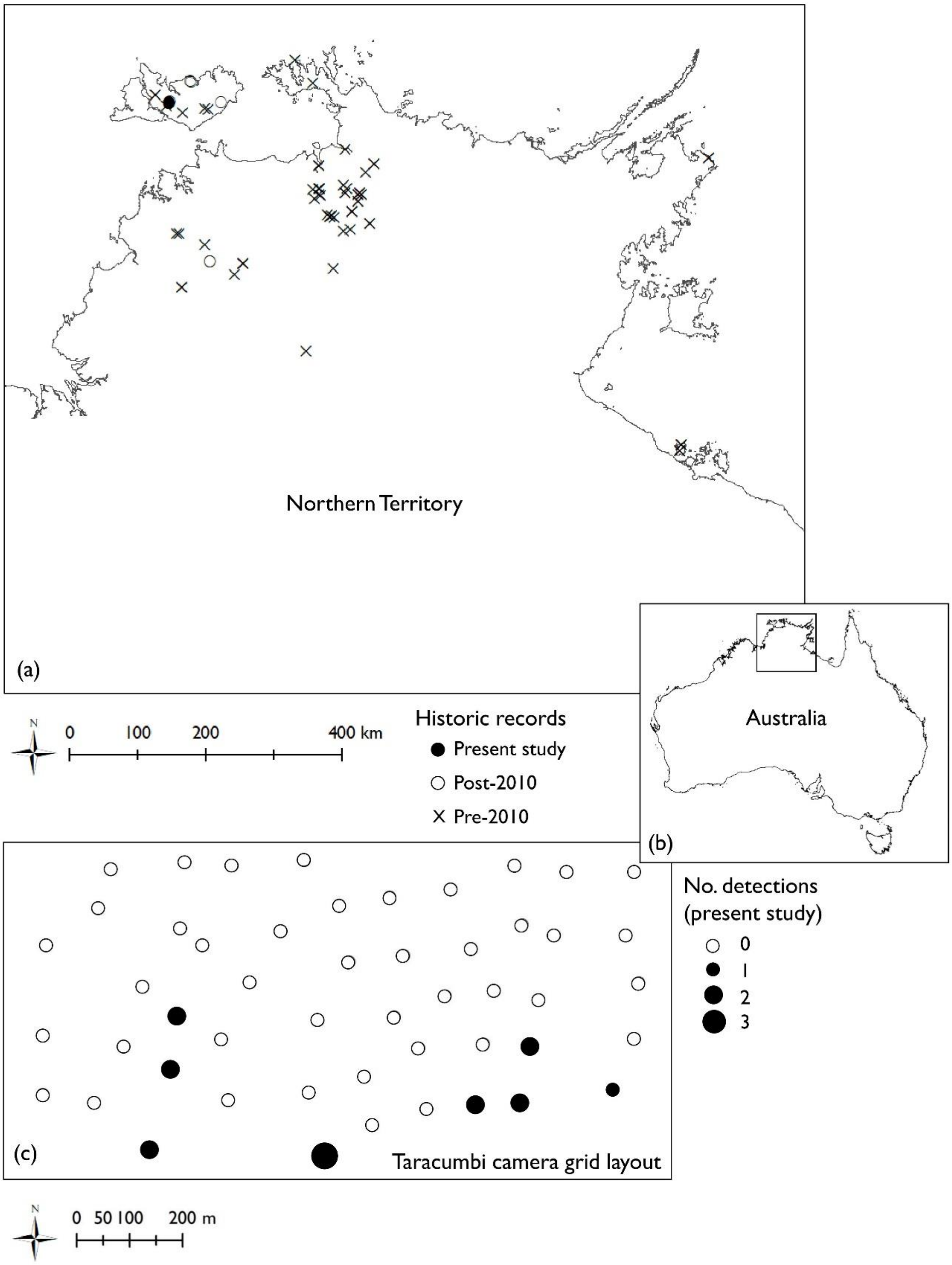

Figure 1. The location of known *Phascogale pirata* records (a) taken from Woinarski *et al.* (2014) and updated to include more recent detections from the Atlas of Living Australia (2018) and unpublished survey data (H. Davies). Crosses indicate records before 2010, open

circles indicate records between 2010 and January 2020, and filled circles indicate records from the present study (and thus the location of the Taracumbi study site); (b) the location of all records relative to Australia; and (c) the approximate layout of motion-sensor cameras at the Taracumbi study site. Open circles indicate that the camera failed to detect *P. pirata*, filled circles indicate a detection event, and increasing size of filled circles indicate an increasing number of detections at a given camera.

Figure 2. Images of *Phascogale pirata*, as captured on remote-sensor cameras on Melville Island, Northern Territory.

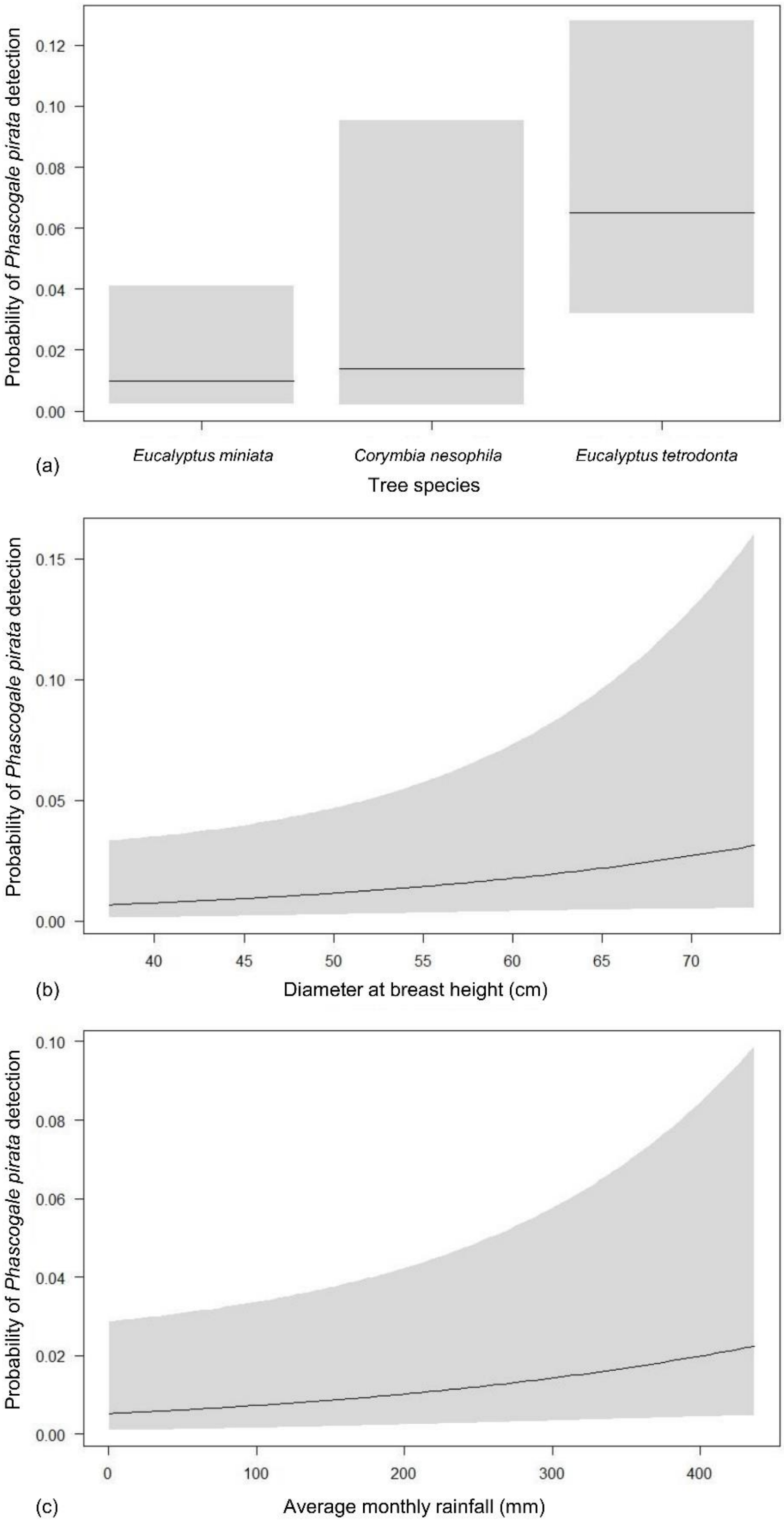

Figure 3. The relationship between the relative probability of detecting *Phascogale pirata* and predictor variables (a) tree species, (b) diameter at breast height, and (c) average monthly rainfall, derived from generalised linear models (GLMs). Each variable relationship is

derived from the best model, while holding other explanatory variables constant (continuous variables at their median and categorical variables at their most common category). Grey bands represent the 95% confidence intervals.

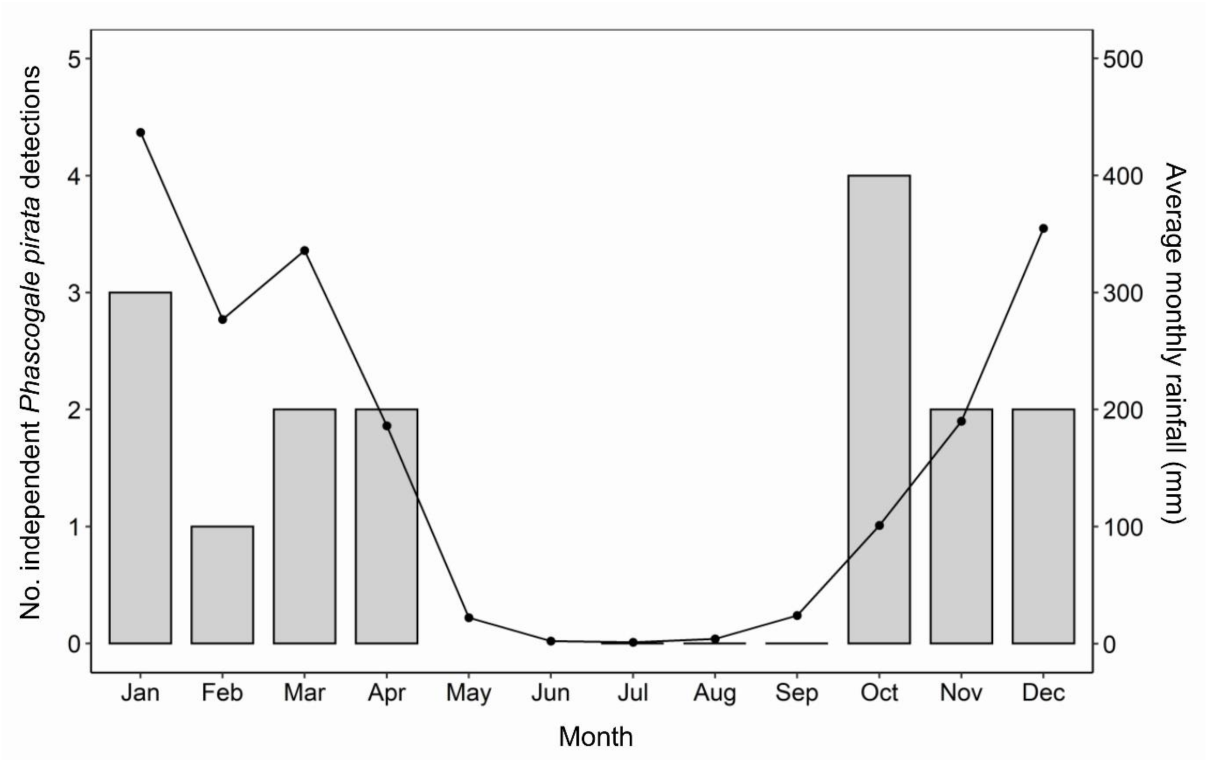

Figure 4. The total number of *Phascogale pirata* detections (bars) and the average monthly rainfall in mm (over the preceding decade, solid line) observed in each month of sampling. Note that data collection began on the 30th of June 2018 and finished on the 4th of May 2019 (and thus the data available for June 2018 and May 2019 are limited).

Table 1. Variables used in generalized linear models (GLMs). Italics indicate name used in reporting of model results. The median value and range is shown for continuous variables, while the most common category is shown for categorical variables.

Variable	Description	Median or most common category	Range
Tree species	Categorical with three levels: *Eucalyptus tetrodonta*, *E. miniata* and *Corymbia nesophilia*	*E. miniata*	n/a
Average monthly rainfall (over preceding decade)	Continuous, standardised. Sourced from Pirlangimpi Airport weather station (Bureau of Meteorology 2019)	188 mm	0.8–437 mm
Diameter at breast height (*DBH*)	Continuous, standardised. Diameter (cm) of main stem measured 1.3 m from the ground	46 cm	37.5–73.5 cm
Season	Categorical with two levels: wet i.e. October–April (average monthly rainfall > 100 mm) and dry, i.e. May–September (average monthly rainfall < 25 mm)	Wet	n/a
Time since *bait* deployment	Continuous, standardised. Scaled from 1 (i.e. within one month of bait replacement) to 7 (i.e. within six months of bait replacement)	3 (i.e. within two months of bait replacement)	1–7

Table 2. Generalised linear model selection results based on Akaike Information Criterion (AIC_c) to test the effects of predictor variables on detection of *Phascogale pirata*, where ΔAIC_c represents the difference between the models AIC_c value and that of the top-ranking model, and W_i is the Akaike weight.

Rank	Model predictors	ΔAIC_c	W_i
1	Average monthly rainfall + DBH + tree species	0.00	0.27
2	Average monthly rainfall + tree species	0.62	0.20
3	DBH + tree species	1.50	0.13
4	Bait + average monthly rainfall + DBH + tree species	1.58	0.12
5	Tree species	2.08	0.09
6	Bait + average monthly rainfall + tree species	2.20	0.09
7	Bait + DBH + tree species	3.44	0.05
8	Bait + tree species	4.01	0.04
9	Average monthly rainfall + DBH	6.05	0.01
10	DBH	7.45	0.01
11	Average monthly rainfall	8.69	0.00
12	Bait + DBH	9.37	0.00
13	Bait + average monthly rainfall	10.27	0.00
14	Bait	11.96	0.00