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Surveys of small and medium sized mammals in northern Queensland with emphasis on improving survey methods for detecting low density populations

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ABSTRACT

The primary aim of this thesis was to investigate the decline of small and medium sized mammals in northern Queensland by: determining the population status of the endangered Bramble Cay melomys, *Melomys rubicola* and investigate the cause of any decline; collating all historical mammal relative abundance data (mammals ≤ 5kg) for the Northern Gulf Management Region (NGMR) with comparisons with other areas in northern Australia where mammal declines have occurred. The secondary aim of the thesis was to investigate methods to improve detection of mammals in northern Queensland by critically determining if camera trapping improved the detection of mammals; and by evaluating the effectiveness of two wax block baits for attracting mammals using camera trapping.

Data collection on Bramble Cay occurred during three survey periods between 2011 and 2014. Survey efforts totalled 1,170 Elliott, 60 camera trap-nights and 10 hours of active searches. To document the variation in cay size and vegetation extent, during each survey, we measured the island area above high tide and vegetated area using GPS.

Mammal relative abundance data were collated for the NG region with a total of 276 1-ha plots surveyed between 2003 and 2016. Of these, 149 plots were surveyed once, 100 plots twice and 27 plots three times. During 2015 and 2016, a further 63 1-hectare plots (52 Cape York and 11 NG) were surveyed with two camera traps integrated into the plot.

In 2016 a bait trail was conducted using two newly developed wax baits (sesame oil infused into bees wax and peanut butter, vanilla essence and oats infused into bees wax) to determine their attractiveness against standard peanut butter bait. Ten transects each consisting of five camera traps placed 50 metres apart were set across the Mareeba Tropical Savanna and Wetland Reserve Nature Refuge (MTSWRNR). Three 21 day trial periods were undertaken with each bait type being rotated using a latin square cross over design. Bait longevity was visually scored for each bait at the end of each trial period.

The three surveys undertaken on Bramble Cay failed to detect any Bramble Cay melomys. The island had experienced a recent, severe reduction in vegetation, which is the primary food resource for the Bramble Cay melomys. Herbaceous cover on the cay decreased from 2.16 ha in 2004 to 0.065 ha in March 2014 before recovering somewhat to 0.19 ha in August–September 2014.

For the NG region, combined datasets from 430 1-ha plots represented sampling efforts totalling 3,316 cage trap nights; 34,480 Elliot trap nights; 6,270 pitfall trap nights and 298.5 hours of spotlighting. A total of 461 individuals, from 24 species were captured. Of the 430 plots, 261 failed to detect any mammals. The majority of the records in the Einasleigh Uplands bioregion were restricted to Blackbraes National Park and a neighbouring property where 80.9% of the plots recorded mammals. Twenty-six (21 native and 5 introduced) mammal species were detected during the surveys in 2015-2016. Of the 732 captures that could be confidently identified, 85.1% (623 camera events from 20 species) were detected by camera trapping, and 14.9% (109 captures from 15 species) were captured using the standardised plot survey method. Camera traps were significantly more effective than the standard plot survey in detecting the dingo, *Canis lupus dingo*, pig *Sus scrofa*, northern quoll, *Dasyurus hallucatus*, northern brown bandicoot, *Isoodon macrourus* and short-beaked echidna, *Tachyglossus aculeatus*. The grassland melomys, *Melomys burtoni*, was detected on significantly more plots using the traditional survey method.

Sampling efforts for the bait trial at MTSWRNR totalled 2,776 camera trap nights (CTN) with 16,531 images and 1269 events collected. A total of 39 species were identified in these events including 13 mammal species. Bait type significantly affected mean camera events for the agile wallaby, *Notamacropus agilis* (P = 0.039). Agile wallabies visited the sesame oil bait stations 1.5 times more than peanut wax bait (P = 0.030). Bait type significantly affected mean species richness of small native mammal species ($P = 0.004$). Peanut butter bait attracted 1.7 times more small native mammal species than sesame oil ($P = 0.003$). There was a significant association between bait type and mould (P = <0.001). Sesame oil bait had the best longevity with no mould present on any of the baits at the end of each trial period.

The Bramble Cay melomys is now extinct on Bramble Cay. The vegetation decline was probably due to ocean inundation resulting from an increased frequency and intensity of weather events producing extreme high-water levels and storm surges, in turn caused by anthropogenic climate change.

The relative abundance and species richness of small to medium sized mammals in NG is considerably lower than other locations in northern Australia where mammal declines have been recorded. Due to a lack of historic data, it is not known if mammals have declined in the region or if they have always persisted at low abundance. An area in the Einasleigh Uplands bioregion appears to have higher abundance of some species and may provide important refugia for some small mammal species. Integrating camera traps into systematic fauna surveys improves the detection of some species in northern Queensland and we recommend integrating camera traps into a monitoring program for mammals at locations which may provide refugia for mammals in the Northern Gulf Management Region. We recommend using standard peanut butter bait for camera trap deployment periods ≤ 21 days. Peanut wax bait may provide a longer life option for camera trapping mammals in northern Queensland for time periods longer than 21 days.

Declaration by author

This thesis *is composed of my original work, and contains* no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

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Publications included in this thesis

Waller, NL, Gynther, IC, Freeman, AB, Lavery, TH & Leung, LK-P 2017, 'The Bramble Cay melomys *Melomys rubicola* (Rodentia: Muridae): a first mammalian extinction caused by human-induced climate change?', *Wildlife Research*, vol. *44*, no. 1, pp. 9-21.

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HAS AUSTRALIA LOST ANOTHER MAMMAL SPECIES? THE STORY OF THE BRAMBLE CAY MELOMYS (*MELOMYS RUBICOLA*)

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The Bramble Cay melomys (*Melomys rubicola*) is endemic to Bramble Cay (Maizub Kaur), a small vegetated sand cay (~4ha) located in the north-east Torres Strait. The species was first noted by Europeans in 1845 when seamen aboard *HMS Bramble* encountered the cay and observed melomys in high densities. The melomys appeared abundant until 1998 when a population of 93.4 (± 35.7 SE) was estimated, based on the capture of 42 individuals. Surveys undertaken in 2002 and 2004 caught 10 and 12 animals, respectively, suggesting the population had declined. Recent surveys in December 2011 and March 2014, both conducted during the green turtle (*Chelonia mydas*) nesting season, failed to capture any individuals. We collated historical and recent data on the population and state of the cay and provide a plausible explanation for the marked decline and possible loss of this species from the island. This information highlights the urgent need to conduct a survey outside turtle nesting season to determine unequivocally the status of the population and, if still required, implement actions to assist in conservation of the species.

Government and Fauna Survey Reports

Gynther, I, Waller, N & Leung, LK-P 2016, *Confirmation of the Extinction of the Bramble Cay Melomys Melomys rubicola on Bramble Cay, Torres Strait. Unpublished Report*, Department of Environment & Heritage Protection, Brisbane, Australia.

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Contributions by others to the thesis

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No works submitted towards another degree have been included in this thesis.

Research Involving Human or Animal Subjects

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LIST OF ABBREVIATIONS

%	Percent
~	Approximately
◦	Degrees
<	Less than
√	Square root
BOM	Bureau of Meteorology
C	Cage trap
CAM	Camera trap
CM	Centimetre
CTN	Camera trap night
DSEWPaC	Department of Sustainability, Environment, Water, Population & Communities
E	Elliott trap
E.G.	For example
EPBC	Environmental Protection and Biodiversity Conservation Act 1999
F	Funnel trap
G	Grams
GPS	Global positioning system
H	Hour
HA	Hectare
IACRC	Invasive Animal Cooperative Research Centre
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
KG	Kilogram
KM	Kilometre
KM^2	Square kilometres
LM	Lumens
M	Metre
MAX	Maximum
MIN	Minimum
MM	Millimetre
N	Number
NG	Northern Gulf
NGMR	Northern Gulf Management Region
NGRMG	Northern Gulf Resource Management Group
P	Pitfall trap
PSMSL	Permanent Service for Mean Sea Level
S	Spotlighting
S.E	Standard error
SP.	Species
SPP.	Species’
T	Traditional survey techniques
TSRA	Torres Strait Regional Authority

CHAPTER 1: INTRODUCTION

1.1 GENERAL INTRODUCTION

Since European settlement, Australia has experienced an alarming decline in native mammalian species. The mammal extinctions, since 1788, currently comprises 30 species. Of these, 29 of are endemic to Australia (McKenzie et al. 2007; Burbidge et al. 2008; Woinarski et al. 2015). Although Australian mammals account for only 6% of the world's mammal species, the extinctions make up approximately 23% of the worldwide mammal extinctions occurring over the past 400 years (Baillie & Groombridge 1996; Fitzsimons et al. 2010).

A recent review of the status of Australian mammals has found that 56 Australian land mammal species meet International Union for Conservation of Nature (IUCN) Red List criteria (20) for listing as threatened and 52 species as near threatened (Woinarski et al. 2015). Alarmingly, 11% of the 273 Australian endemic land mammal species are extinct, 21% are threatened and 15% near threatened (Woinarski et al. 2015). Many of the threatened and near threatened species continue to decline, with some monitoring programs indicating population reductions of up to 90% over the past two decades (Woinarski et al. 2015).

Most Australian mammalian extinctions and population declines since European settlement have occurred in the arid, semi-arid and temperate regions of Australia (Fisher et al. 2013; Woinarski et al. 2015). Offshore islands, coastal areas and tropical northern Australia have in the past provided some refuge to mammal communities (Short & Smith 1994). There is clear evidence that recently, however, regional extinctions and reduced distributions (some as much as 10% of their former range) of small and medium sized mammals have been occurring across parts of tropical northern Australia (Fitzsimons et al. 2010; Woinarski et al. 2010; Woinarski et al. 2011a; Ziembicki et al. 2015). These declines do not appear to be limited to mainland Australia as a number of mammal populations have become extinct from islands on which they formerly occurred, presumably as a result of threats introduced from the mainland such as feral cats and cane toads (Woinarski et al. 2011b; Woinarski et al. 2015). The timing and location of mammal decline across Australia appear to coincide with the spread of human settlement and introduced species the cat, *Felis catus*, and red fox, *Vulpes vulpes* (Woinarski et al. 2015). The timing may also be linked to a loss of indigenous land management, in particular, indigenous fire regimes (Woinarski et al. 2015).

The decline of northern Australian mammals appears to follow patterns of decline in other regions of Australia where the extinction prone species appear to be the larger more specialised species such as large rodents and marsupial species (e.g., black-footed tree-rats, bandicoots, possums and quolls) (Woinarski et al. 2007; Woinarski et al. 2015). The smaller more generalist species that favour disturbance appear to increase in numbers (Woinarski et al. 2007).

The mammal declines in northern Australia are largely attributed to a combination of environmental factors, including fire regime, the grazing of introduced herbivores, drought, predation by cats and wild dogs and exotic wildlife disease (Woinarski et al. 2011a; Kutt 2012; Woinarski et al. 2015). Identifying the principle cause, however, is particularly difficult for several reasons:

1. Historical declines occurred simultaneously with many changes to Australian ecosystems, including changes to land use such as pastoralism, fire regimes, the introduction of exotic fauna and flora species and potentially exotic wildlife diseases (Woinarski et al. 2011a; Fisher et al. 2013);
2. current land use patterns and fire regimes are largely discontinuous and vary geographically and, in their scale, and intensity (Burbidge et al. 2008);
3. species react inconsistently to threats, or different combinations of threats, and individual species may vary in their responses to individual threat factors at different sites and times (Woinarski et al. 2015); and
4. many of the mammals found in tropical savanna are at critically low densities that experimentally evaluating factors influencing populations is not logistically possible.

There is also an absence of reliable data on the past and present distributions of Australian mammals (McKenzie et al. 2007; Burbidge et al. 2008) and few long-term monitoring sites have been established in Northern Australia to track changes in mammal populations. The decline of small and medium sized mammals in northern Australia has largely been documented in the Northern Territory, where systematic repeated sampling has occurred over more than 20 years (Woinarski et al. 2010;

Woinarski et al. 2011a; Woinarski et al. 2011b). In northern Queensland, however, it is impossible to determine if these mammals have declined as there are no long-term historical data that has been systematically collected. Fauna survey data available for northern Queensland, however, do indicate that small and medium sized mammals are at very low diversity and abundance in many locations (Winter & Lethbridge 1995; Preece 2009, 2010; Winter 2010; Preece 2011; Preece & Preece 2012; Preece 2012; Preece & Franklin 2013; Starr & Waller 2013b, 2014b, 2014a).

This thesis focuses on small and medium sized (≤5kg) terrestrial mammals found in northern Queensland across three Natural Resource Management regions, Northern Gulf, Cape York and Torres Strait. These regions are the least studied in northern Queensland due to their remoteness. Hereafter, reference to Northern Gulf, Cape York and Torres Strait refers to the Management Region unless stated otherwise. The Wet Tropics Management Region is excluded from this study as this management region has been more extensively studied and differs quite differently in vegetation and mammal communities than other parts of northern Australia (Ziembicki et al. 2015).

1.2 SMALL AND MEDIUM SIZED MAMMALS IN NORTHERN QUEENSLAND

A review of Menkhorst and Knight (2011) identified a total of 61 species of small and medium sized terrestrial mammal occurring across the three management regions. These include ten macropods, 19 rodents, 12 dasyurids, four bandicoots, nine possums, six gliders and one monotreme species. Of the 61 species, three are listed as endangered, two near threatened and one vulnerable according to Queensland Government legislation *(Nature Conservation Act 1992).* The three species listed as endangered include:

- the Bramble Cay melomys, *Melomys rubicola* a small rodent species endemic to a small sand cay in north-eastern Torres Strait;
- the northern bettong, *Bettongia tropica* a small macropod endemic to north-eastern Queensland, occurring in only three locations; and
- the spotted-tailed quoll, *Dasyurus maculatus gracilis* a subspecies of the spotted-tailed quoll found in north-eastern Queensland, occurring in two discrete subpopulations.

Sixteen of the species found in northern Queensland that are listed as least concern under Queensland Government legislation are considered to be currently declining (Woinarski et al. 2014). These species either occupy a large geographical area or are not considered to be declining at a rate fast enough to be listed as a conservation priority.

1.3 THREATS TO MAMMALS IN NORTHERN QUEENSLAND

The most widespread processes considered to be causing threats to small and medium sized mammals in Northern Queensland are the impacts from extensive land use (such as grazing), changed fire regimes and cat predation, possibly coupled with disease. Modern fire regimes, particularly frequent and extensive late dry season fires, and overgrazing by livestock, have been implicated as the major factors causing the recent observed decline of many northern Australian small mammal species (Burbidge & McKenzie 1989; Russell-Smith et al. 2001; Woinarski et al. 2007; Yates et al. 2008; Legge et al. 2011; Woinarski et al. 2011a; Woinarski et al. 2015). The exception to this is the Bramble Cay melomys which is considered declining due to the species being restricted to a small isolated sand cay that is vulnerable to erosion and inundation. The following provides an overview of the principle threats to small and medium sized mammals in northern Queensland.

1.3.1 Grazing

Northern Australia is considered largely intact due to large areas of continuous open woodlands. Although the original condition of the vegetation is unknown, recent surveys strongly suggest that a major cattle-driven transformation has taken place in the landscape (Foran 1980). Approximately 90% of Northern Gulf and 50% of Cape York is currently under pastoral lease (Gobius et al. 2014). Historically cattle numbers across Northern Gulf and Cape York were low; however, since the 1960's the numbers have continued to increase. Growth in stock numbers has been influenced by factors that have reduced previously high mortality rates, in part made possible by improved road access, introduction of drought and tick resistant breeds, better management through fencing and more water points, and increased use of feed supplements (Crowley 2015). Currently there is no cattle grazing occurring in the Torres Strait and this region, therefore, will not be included in the discussion in this section.

Grazing by introduced herbivores can indirectly affect terrestrial mammals in several ways. Grazing may directly reduce food availability to native herbivorous and granivorous species through changes to flora composition and abundance (Sharp & Whittaker 2003; Kutt & Fisher 2011; Legge et al. 2011). Trampling causes soil compaction which affects the habitat quality for ground-dwelling and fossorial animals (Legge et al. 2011; Woinarski et al. 2014). Cattle grazing can also facilitate or suppress recruitment of woody vegetation (Harrington 1991; Archer 1995; Scanlan et al. 1996; Van Auken 2000; Burrows et al. 2002) and, therefore, indirectly influence species reliant on that vegetation. Reduction in ground cover caused by grazing may also increase the risk of predation to some species (Fleischner 1994; Legge et al. 2011).

In Northern Australia it is difficult to experimentally evaluate the impacts of grazing on small and medium sized mammals as most of the landscape has been grazed for a long period. Mammals in many places are at low densities and fire frequencies vary considerably across the landscape. Studies that have sampled grazed and previously grazed (destocked) land have shown recovery or higher abundance of some species of mammal in destocked habitats (Woinarski & Ash 2002; Kutt & Woinarski 2007; Read & Cunningham 2010; Legge et al. 2011; Kutt & Gordon 2012). One study undertaken in northern Queensland, however, found that after 5 years of exclusion, mammals remained at low abundance (Kutt et al. 2012). Furthermore, a survey undertaken by Northern Gulf Resource Management Group (Starr & Waller 2014a) in the Einasleigh Uplands bioregion, found that even after 15 years of cattle exclusion, small and medium sized terrestrial mammals remained at low diversity and abundance. Although grazing may negatively impact on disturbance prone species, some species may prefer disturbance caused by cattle grazing (Woinarski & Ash 2002; Read & Cunningham 2010), thus a mix of low stocking rates and destocking of some grazed areas, can provide habitat that is suitable for a range of species. Rocky habitats are naturally protected from grazing impacts due to their inaccessibility and may provide refuge for many fauna species impacted by cattle grazing (Fisher *et al.* 2004).

1.3.2 Fire

Australian tropical savanna is a frequently burnt landscape, dominated by large areas burnt annually with few escaping fire for more than five years (Dyer et al. 2001a; Woinarski et al. 2007; Yates et al. 2008). Since European settlement, indigenous land management has been lost from many parts of northern Australia and therefore fire regimes have largely changed. Since European settlement fire management across Cape York Peninsula has changed from traditional burning throughout the dry season to pastoralists burning predominantly in the early dry season (no deliberate burns after August) (Crowley & Garnett 2000). This change in fire management is aimed to burn with cool, patchy fires over a small spatial area as the moisture content in the vegetation limits burning (Liedloff et al. 2001; Russell-Smith et al. 2003; Woinarski et al. 2007). In contrast, the late dry season fires are hotter and more extensive due to the dry vegetation providing an ignitable fuel (Woinarski et al. 2007). Throughout the Northern Gulf, fires are less frequent as a result of lower annual rainfall and more intensive grazing which reduces fuel loads (Kutt et al. 2012a). Many graziers across the Einasleigh Uplands bioregion do not use fire regularly as part of their property management as they consider burning grasses a waste of feed that can be used for cattle (Crowley 2015). Islands across the Torres Strait have largely maintained traditional fire regimes due to continuing indigenous management. Some of the vegetation and fauna communities found in this region are therefore some of the best examples of biodiversity found in northern Queensland, particularly when compared to similar habitats in Cape York (3D Environmental 2012).

Fire can have detrimental impacts on fauna communities, particularly if they are of high intensity over extensive areas or occur frequently. Some fauna are particularly vulnerable to extensive and frequent fires (Russell-Smith et al. 2001; O'Malley 2006; Legge et al. 2008; Andersen et al. 2012; Valentine et al. 2012), especially species that utilise food and den resources that are reduced by fire, or have small home ranges (Yates et al. 2008). Large, extensive and intense fires have been implicated as the cause of decline for small and medium sized mammals in northern Australia (Woinarski et al. 2004; Andersen et al. 2005; Woinarski et al. 2010; Woinarski et al. 2015). Extensive and frequent fires reduce resource availability to mammal species by causing widespread reduction in seeds and grasses, mortality of small trees and shrubs, and reduce flowering and fruiting of savanna trees, particularly in the season following the fire (Liedloff et al. 2001; Russell-Smith et al. 2001). Frequent fires may have long-term consequences for mammal species that prefer unburnt habitat, by reducing the abundance of woody understorey and mid-storey vegetation (Woinarski & Ash 2002) which leads to decreased availability of food resources and den sites. There is some archaeological evidence that fire has reduced the abundance of native fruit producing plants in some areas since European settlement (Atchison 2009). Species such as the black-footed tree rat, *Mesembriomys gouldii*, and common brushtail possum, *Trichosurus vulpecula*, have been found to strongly prefer habitat that is long unburnt, suggesting that high fire frequency is detrimental to these species. (Kerle & Burgman 1984; Kerle 1985; Woinarski et al. 2004).

Several authors have implicated the possible link between extensive and frequent fires and small mammal declines in northern Australia (Fitzsimons et al. 2010; Woinarski et al. 2011a; Radford 2012; Lawes et al. 2015). This link was recently demonstrated by Lawes et al. (2015), where fire extent was found to be the most detrimental factor in small mammal declines in Kakadu National Park. Small mammals appeared to be resilient to fires in areas less than 10 hectares but declined dramatically in abundance and species richness when the surrounding area (less than 3 kilometres) was subjected to repeated large fires (Lawes et al. 2015). This is presumably due to small mammals not having the ability to disperse over large distances to unburnt habitat to seek refuge (Lawes et al. 2015). Some species are surprisingly resilient to fire such as the delicate mouse, *Pseudomys delicatulus*, which, after an extensive, high intensity fire, appeared unaffected, while other species appeared lower in abundance (~90% reduction) in burnt than unburnt patches (Legge et al. 2008). The author considered the resilience of this species to fire was due to its small size and ability to avoid post-fire predation by hiding in small crevices (Legge et al. 2008). There is some evidence that although some small-mammal species in northern Australia initially survive extensive, high intensity fires by seeking refuge in burrows or rock crevices, they decline in abundance during the months after the fire (Legge et al. 2008; Lawes et al. 2015). This is due to the indirect effects of fire on survival and reproductive output due to a reduction in food supply and increased exposure to predation. This is likely exacerbated by high fire frequency which further limits population recovery after fire (Legge et al. 2008; Andersen et al. 2012; Lawes et al. 2015).

Large-scale high intensity wildfires late in the dry season are more likely to occur in unburnt habitat because of the greater fuel loads. Intense wildfires, at times reaching the crowns of trees, have devastated large areas of the Northern Gulf region, causing losses to stock and wildlife (NGRMG 2013). Unfortunately, high intensity wild fires are predicted to occur more frequently in Northern Australia as a result of climate change. Ideal fire regimes for conserving biodiversity and preventing high intensity wildfires include a mosaic of small fire patches across the landscape, preferably conducted in the early dry season (Dyer et al. 2001b). Fire regimes that maintain some long-unburnt areas are important for species such as the common brushtail possum and black-footed tree-rat (Woinarski et al. 2007). Low intensity mosaic fire regimes aim to prevent the build-up of high fuel loads that may contribute to intense wildfires later in the dry season and provide a mosaic of habitat patches where fauna can disperse and seek refuge after the fire. Furthermore, patch burning can be used to promote mammal species richness by increasing the diversity of habitat types or post-fire successional stages in the landscape (Russell-Smith et al. 2001; Fisher et al. 2004). Traditional fire management, such as burning regular, small fires, lit throughout the year, may be used to increase the diversity of flora to benefit a greater range of species. This fire regime has been successful in maintaining a high representation of fire sensitive vegetation types, high diversity of vertebrate fauna (including rare and range-restricted species), and maintaining relatively low numbers of exotic plant species (Yibarbuk et al. 2001)

1.3.3 Predation by cats and feral dogs (dingoes)

The pest species which poses the most serious threat to small and medium sized mammals in northern Australia is the feral cat, *Felis catus*, (IACRC 2007). The species is listed by the Queensland government as ‘abundant and widespread’ throughout the Northern Gulf and Cape York region (IACRC 2007). The species is also found on many inhabited and uninhabited islands across the Torres Strait. The impacts of cats on biodiversity values have been listed as key threatening processes under the Australian Government’s *Environment Protection and Biodiversity*

Conservation Act 1999 (EPBC). The feral cat has been implicated in the extinction of 19 mammal species, as well as recent declines of mammals across northern Australia (Woinarski et al. 2011a; Fisher et al. 2013; Ziembicki et al. 2015).

The success of the feral cat can be largely attributed to the lack of native meso-predators and the efficacy of the species hunting behaviour (Kutt 2012). The group of species most impacted by feral cat predation in northern Australia are critical weight-range mammals weighing from 35 g to 5.5 kg (Dickman 1996; Kutt 2012).

The range of species consumed by feral cats varies depending on the environment and prey availability. Kutt (2012) examined a total of 169 cat stomachs (974 items) in northern Queensland and found mammals were the dominant prey item by mass (100-3500 g range) comprising 22% of the prey items identified and represented by 23 species. Kutt (2012) also demonstrated that feral cats consume a large amount of prey per cat (approximately 200 g per cat, per night) and that mammals are the dominant prey item by mass in the 100-3500 g range (other prey species have few species within this size class), although reptiles and birds within 10 g-100 g weight range are also a major component of the diet. The size and dietary mass of native fauna in the cat diets revealed by the author are of particular concern for northern Australia, suggesting that predation by feral cats is possibly a significant cause of local mammal extinctions in the small size range.

In northern Queensland, 45 of the 61 small and medium sized mammal species known to occur across the three NRM regions are within the weight range preferred by feral cats in northern Queensland (100 – 3500 g) (Menkhorst & Knight 2011; Kutt 2012). Nine of the 16 species that fall above or below this size class are considered to be likely prone to predation by feral cats due to the species ecology (particularly foraging on the ground) or their young being within the preferred size range (Woinarski et al. 2014). Furthermore, sixteen of the 61 species are considered to be declining according to the ‘action plan for Australian mammals 2012’ (Woinarski et al. 2014) and all of these overlap with the medium to high prey size selectivity for the feral cat diet recorded in north Queensland by Kutt (2012). The exception to this is the Bramble Cay melomys, which is found on an isolated sand cay where feral cats have not been recorded (Latch 2008). Cats are however, listed as a potential threat to the species if cats were introduced to the Cay (Latch 2008).

Manipulative experiments have been used to critically determine the response of small mammal populations to the removal of exotic predators. A South Australian study was successful in sustaining abundant populations of several small mammal species in predator excluded areas, however these areas also excluded the European rabbit, an introduced herbivore (Moseby et al. 2009). Limited research on the benefits of cat exclusion for small mammal populations has been undertaken in northern Australia. The population of long-haired rat, *Rattus villosissimus,* was maintained at sites where cats were excluded, while control plots the species was extirpated (rapidly in one plot, gradually in another) by cat predation (Frank et al. 2014). Results from a recent study on Melville Island, in the Northern Territory, suggest that feral cats are driving the brush-tailed rabbit-rat, *Conilurus penicillatus*, to extinction on the island and are likely a significant driver of the species decline across Northern Australia (Davies et al. 2017).

Control options for feral cats include fencing (the only feasible option for complete removal from protected areas), shooting, poisoning using lethal baits and trapping. Currently there are no viable options for controlling feral cats over large areas. Baits specifically developed to target cats (Eradicat® and Curiosity®) have shown to be effective; however, these may also poison non-target species (Hetherington et al. 2007; Johnston et al. 2011). More research is required to further develop these baits for safe application across Australia.

There is some evidence that dingoes and wild dogs may impact on native mammals through predation and by competition with native predators. Although dingoes in some areas have shown a preference for mammals in their diets (Dickman 1996), their impacts on mammal communities are considered to be less than that of feral cats (Bradshaw et al. 2007). Dingoes and wild dogs are considered to have some beneficial impacts on native mammals, by regulating feral cats and foxes (Dickman 1996). However, this tenet is not fully understood and is subject of considerable debate recently (eg. Ritchie *et al.* 2014; Allen *et al.* 2014; Letnic *et al.* 2012). The risk of predation of livestock has resulted in wide scale control of feral dogs across northern Queensland, using toxic baits and shooting by property managers (Woinarski et al. 2015). The subsequent reduction in feral dog numbers is considered to be one of the key reasons why cat populations have become widespread and subsequently caused a loss of small native mammals (Fitzsimons et al. 2010; Brook & Kutt 2011; Woinarski et al. 2015).

1.3.4 Cane toads

Although the introduced cane toad, *Rhinella marina*, has been implicated as a principle factor in the decline of small and medium sized mammals in northern Australia, only two mammal species, the northern quoll and spotted-tail quoll, appears to have undergone serious declines that can be linked to this pest (Burnett 1997; Ziembicki et al. 2015). Northern quoll populations are reduced due to increased mortality resulting from cane toad poisoning during predation attempts (O'Donnell et al. 2010; Woinarski et al. 2015). The northern quoll has declined rapidly and extensively, and continues to do so, across northern Australia following the introduction of the cane toad (Burnett 1997; Woinarski et al. 2015). Some quoll populations in northern Queensland are recovering from toad invasion, possibly due to the species learning prey avoidance (Woinarski et al. 2008; Ziembicki et al. 2015). A recent camera trapping survey undertaken at South Endeavour Nature Refuge (June 2016) found evidence that wild northern quolls were avoiding cane toads (Starr et al. 2015b). Cane toads have only been found on four islands in the Torres Strait (Thursday, Horn, Hammond and Murray). Only one invading individual was found on Murray island and was removed before a population was established (N. Waller, unpublished data). Cane toads are likely to only have minimal impacts to mammals on Torres Strait islands as the northern quoll is not found in the region.

1.3.5 Disease

Disease is implicated as a potential factor in the decline of mammals in northern Australia (Woinarski et al. 2011a; Woinarski et al. 2015; Ziembicki et al. 2015). However, no link has yet been made (Reiss et al. 2015). There is very little knowledge of infectious diseases in wildlife in Australia and even less known about the prevalence of disease in northern Australian mammals.

Disease may cause mammal population declines through increased mortality and morbidity and reduced fecundity. For example, in Tasmania, Devil facial tumour disease has caused a substantial decline in the Tasmanian devils, *Sarcophilus harrisii*, with over 50% of some populations infected with the disease (McCallum & Jones 2006). Toxoplasmosis has been implicated as a potential disease problem due to cats being a host and vector for the disease (Hollings et al. 2013). Toxoplasmosis can change mammal behaviour, such as causing increased docility and activity during daylight hours. Animals showing these behaviours may therefore be more prone to predation.

Reiss et al. (2015) investigated disease prevalence and small mammal declines in northern Australia. Disease prevalence was investigated in four species of mammal (brushtail possum, northern brown bandicoot, northern quoll and brush-tailed rabbit-rat) in the Northern Territory. Each species was chosen as a representative of broad taxonomic groups that are considered to be declining in the top end. A total of 281 anaesthetised individuals and 113 other passive samples were assessed for disease prevalence. Of the 281 individuals anaesthetised 94 % were considered to be in normal health, and, of the 17 considered to be suffering from disease, eight were suffering from acute disease (e.g. Injuries). Several pathogens were identified in mammals during the study (including mammalian herpesvirus, enteric *Salmonella* spp., protozoal haemoparasites (trypanosomes, *Babesia* spp., *Hepatozoon* spp.), enteric protozoa (*Cryptosporidium* spp. and *Giardia* spp.), microfilaria and *Toxoplasma*). Although some of these pathogens are considered capable of impacting on population health, the author found no evidence that the pathogens are responsible for, or a risk factor in the decline in mammals in the Northern Territory (Reiss et al. 2015).

1.4 ISLAND REFUGES?

Islands have provided refuge for some mammal species in Australia. However, in other cases they may act as 'death traps' if mainland threats extend onto the island (Woinarski et al. 2011b; Woinarski et al. 2015). The main difference between threats to mammals on the mainland and islands in Australia is that some islands remain free from non-native predators, invasive rodent species and cane toads (Woinarski et al. 2015; Ziembicki et al. 2015). An association has been found between introduced predators and mammal extinctions from islands in Australia (Burbidge &

Manly 2002). Extinctions tend to occur more frequently on arid islands and those without refuge habitats such as rock piles. Further, species that are restricted to the ground and are relatively large are more prone to extinction (Burbidge & Manly 2002). Eradication of invasive species is possible on small islands. However, is almost impossible on larger islands or continents (O'Donnell et al. 2010; Ziembicki et al. 2015). Cats and black rats, *Rattus rattus*, are found on a considerable number of inhabited and uninhabited islands in the Torres Strait. Recently cane toads have become established on some islands as discussed above (Lavery et al. 2012).

Mammal fauna in northern Australia appear to remain relatively intact on some islands such as Groote Eylandt, Tiwi islands and some islands in Torres Strait. Many of the islands in Northern Territory and Western Australia support threatened species, such as the northern quoll, golden bandicoot, *Isoodon auratus*, and black-footed tree rat (Ziembicki et al. 2015). Inventories of small and medium sized terrestrial mammal species in the Torres Strait, comprise for the most part common species that are found on the mainland, with the exception of the ‘Endangered’ Bramble Cay melomys. The water mouse, *Xeromys myoides*, which is listed as threated in the *Nature Conservation Act 1992,* may be found on some western Torres Strait islands as it is found in similar habitat in Papua New Guinea (RPS 2008a).

1.5 FAUNA SURVEYS IN NORTHERN QUEENSLAND

1.5.1 Northern Gulf Management Region

Northern Gulf remains one of the least studied management areas for biodiversity in Australia. There are few published reports specific to the region, and numerous unpublished reports (Preece 2009, 2010; Winter 2010; Preece 2011; Vanderduys & Kutt 2011; Preece & Preece 2012; Preece 2012; Vanderduys et al. 2012; Preece & Franklin 2013). Systematic terrestrial vertebrate fauna surveys have been conducted on only 33 properties across the Northern Gulf region, and only 8 properties have had these surveys repeated. Some single species studies have occurred, mostly on threatened species such as the northern quoll and northern bettong (Johnson & McIlwee 1997; Laurance 1997; Vernes & Haydon 2001; Vernes & Pope 2001;

Woinarski et al. 2008). There appears to be no historical survey data, and the oldest fauna survey records date back as far as 2003.

Of the surveys that have been conducted, only a handful reported reasonable diversity and abundance of small and medium sized mammals. On those sites that mammals were found, there appears to be a relationship between land use and altitude, with small and medium sized mammals occurring more often on conservation lands at higher altitudes (Vanderduys & Kutt 2011; Vanderduys et al. 2012; Starr & Waller 2014a) or in moist habitats (Burnett 2001). This highlights that these areas are likely to be important refugia for this mammalian size class in northern Queensland. The majority of surveys conducted in this region, found either very low diversity and abundance of small mammals, recorded unstable small mammal populations or detected no small mammals during the survey (Preece 2009, 2010, 2011; Sanders 2011; Smith et al. 2011; Kutt et al. 2012b; Preece & Preece 2012; Preece 2012; Preece & Franklin 2013; Starr & Waller 2014b).

The abundance of small and medium sized mammals in Northern Gulf appears comparable to more recent survey results from Kakadu National Park where small mammals were found to be declining. The abundance of small mammals in Northern Gulf was low even in 2003 when reasonable numbers of small mammals were still being captured in Kakadu National Park. The survey data collected from 2003 to 2013, do not show any small mammals on 63% of sites surveyed across the Northern Gulf region between 2004 and 2014, suggesting either very low abundance or complete absence. In the Northern Gulf region. It is impossible to determine if small-medium sized mammals have declined due to there being no long-term historical data.

1.5.2 Cape York Management Region

Although Cape York Peninsula has been more extensively studied than the Northern Gulf and Torres Strait regions, there have still been very few comprehensive fauna surveys undertaken historically across the region. Many of the historic data is specific to species or taxonomic groups such as birds, reptiles or amphibians (Covacevich et al. 1982; Covacevich 1987; Cameron & Cogger 1992; Pritchard et al. 2009; Perry et al. 2011). The first systematic fauna surveys (30 sites) were undertaken in 1980-1 by Winter & Atherton (1985) in the vicinity of Weipa. Most subsequent surveys in the next 20 years were undertaken using inconsistent sampling methodologies and, although these surveys may provide good baseline sampling, the surveys are not easily repeated or compared across the landscape. Since 2008 over 250 sites have been surveyed across Cape York using standard methodologies recommended by Eyre et al. (2012) for Sampling terrestrial vertebrates in Queensland (Preece & Preece 2012; Starr & Waller 2013b). Although this number of sites appears to be considerable, large information gaps, particularly for the state of mammals on Cape York Peninsula still remain. Further, due to the paucity of historical data, it is impossible to determine if small and medium sized mammals have declined across Cape York Peninsula.

An exception to this is the long-term data of various sites surveying for the common brushtail and common ringtail possum, *Pseudocheirus peregrinus*, that has been collected by Winter (2007). The data indicated that north Queensland brushtail possum populations undergo large fluctuations in population numbers, at times low enough that spotlighting surveys have failed to detect a single individual. Winter's study also indicated that over the past 30 years the species suffered significant localised population declines across some areas of Cape York Peninsula (Winter 2004; Winter et al. 2004; Winter 2007).

1.5.3 Torres Strait

Due to the remoteness of the Torres Strait region, there is very limited fauna survey data and most historical records are only found in museum archives. Prior to 2007 there has been no systematic surveys or population monitoring of mammals, with exception of the Bramble Cay melomys, where the population has been loosely monitored since 1998 (Dennis & Storch 1998; Latch 2008; Dennis 2012). Between 2008 and 2009 systematic fauna surveys were undertaken on 15 inhabited islands (RPS 2008a, 2008b, 2008c, 2008e, 2008f, 2008g, 2008d, 2009b, 2009f, 2009a, 2009c, 2009h, 2009e, 2009g, 2009d). These surveys were rapid surveys undertaken over four nights at each island, aimed at collecting species inventories on each island, by targeting prominent habitats on each island. It is likely that some small mammal species have remained undetected by these surveys as they were undertaken for a short duration with limited trap numbers. Further, anecdotal evidence of a striped mammal was recorded on Masig, and may have been detected during the survey, the author suggested the species may be extinct (RPS 2008f).

The results of fauna surveys across Torres Strait indicate that terrestrial mammal richness is low and is correlated with island size, isolation and the diversity of habitat types (Lavery et al. 2012). Although terrestrial mammal richness was low, species such as the grassland melomys, *Melomys burtoni*, were found at particularly high abundances in grasslands and vine forests, indicating the populations are in good health (RPS 2008a, 2008c, 2009a, 2009c, 2009d), in the future these islands may become strongholds for some small and medium sized terrestrial mammals, if populations decline on the mainland.

There are currently 8 species of small and medium sized terrestrial mammal found in the Torres Strait, and 7 of these are listed as least concern. The Bramble Cay melomys is one exception, the species is considered to have declined since the original population surveys in 1998 (Latch 2008). Major threats to the Bramble Cay melomys are recognised as being erosion of the cay, introduced plants, predators, competitors and disease, as well as genetic drift and inbreeding depression (Latch 2008). In the face of such threats, both *The Action Plan for Australian Rodents* (Lee 1995) and the national recovery plan (Latch 2008) emphasised the importance of establishing a monitoring program for the Bramble Cay melomys. However, other than the 1998, 2002 and 2004 censuses (Dennis & Storch 1998; Latch 2008), no further monitoring or implementation of the additional recommended recovery actions have occurred. As this species is restricted to a small isolated sand cay and has a small population size that declined at a significant rate between 1998 and 2004, with little trend information published subsequently, further population monitoring is critical in determining the current conservation status of the species.

1.6 CAMERA TRAPPING, THE WAY FORWARD?

Given the paucity of historical data in northern Queensland, it is difficult to determine if small and medium sized mammals have declined or if they have always persisted at low abundance in the landscape. Systematic fauna surveys are time consuming and financially expensive to undertake, and there are large areas that have yet to be surveyed. Further, funding for fauna surveys is often difficult to obtain, particularly for areas that have been surveyed in the past. Many of the recent fauna surveys have therefore been undertaken in areas that have never been surveyed in an attempt to collect fauna inventories. Unfortunately, small and medium sized mammals are at such low abundance they are difficult to detect using traditional fauna survey methodologies, and in some cases only one individual is detected during a survey period.

Camera traps are increasingly being used in Australia to survey fauna populations as they offer many advantages over traditional fauna survey techniques that rely on live-capture of animals. Camera traps are able to be deployed for extended time periods (up to several months) without intervention, making this method particularly useful for remote areas (Gillespie et al. 2015). Camera trapping may increase the detection of cryptic species that are difficult to capture in live traps and provides a more ethical option, particularly for fauna that may be injured in traps or suffer from stress related myopathy (such as small macropods) (Claridge et al. 2010).

Several studies have proven the effectiveness of camera trapping for detecting small and medium sized mammals in Australia, particularly when camera traps are deployed for longer periods (Claridge et al. 2010; De Bondi et al. 2010; Paull et al. 2011; Paull et al. 2012; Swan et al. 2014). Remote cameras have also been found to be as or more effective at detecting target mammal species when compared to more traditional fauna survey techniques (De Bondi et al. 2010; Paull et al. 2012; Swan et al. 2014). When accounting for cost, camera traps have been found to be more cost effective than live trapping (up to nine times cheaper when excluding the initial cost of the camera traps) and therefore funds saved can be invested into increasing the sampling effort across larger spatial or temporal scales (De Bondi et al. 2010; Meek et al. 2012). Remote cameras are also a highly accessible tool for indigenous ranger groups due to the ease of operation.

Camera traps, however, may have some limitations for sampling populations of small and medium sized mammals in northern Queensland, particularly species that do not have distinctive taxonomic features easily seen using camera trap imagery. Some species may only be identified to genus or family level due to taxonomic differences being only differentiated by close examination of the animal (Frost 2008; Menkhorst

& Knight 2011; Meek et al. 2012). Remote cameras may also fail to detect some species that are easily captured using traditional fauna survey methods, such as Elliott or cage trapping (Swan et al. 2014). Therefore, using a combination of camera trapping and traditional fauna survey methods may be the key to improving detection rates for a range of small and medium sized mammals (Swan et al. 2014).

In northern Queensland the use of camera traps in environmental monitoring programs has been limited and the benefits of using cameras for surveying small and medium sized mammals in northern Queensland has not been studied.

1.7 THESIS AIMS

The primary aim of this thesis was to investigate the apparent decline of small and medium sized mammals in northern Queensland. In Chapter two I focus on the endangered Bramble Cay melomys, a rodent endemic to a small isolated sand cay in the Torres Strait. Surveys were undertaken to investigate the decline of the species and factors influencing this decline. When two surveys failed to locate any individuals, a third survey was undertaken to recover any remaining individuals for a captive insurance population. Due to the species small population size, camera trapping was integrated into the third survey in an attempt to improve the chance of detecting the species.

In chapter three I aimed to collate all historical mammal population data for the Northern Gulf natural resource management region. Several fauna surveys were undertaken across the region to supplement this data. We calculated the relative abundance for each species of mammal captured in Northern Gulf (for species up to five kilograms in body weight) and compare the capture rates to other regions in northern Australia where declines in some small and medium sized mammals have been recorded. We integrated camera traps into the systematic plot-based survey method, which is widely used across northern Australia to critically determine if camera trapping improved the detection of small and medium sized mammals in northern Queensland.

In Chapter four I aimed to evaluate the effectiveness of two wax block baits in camera trapping small and medium sized mammals in tropical savanna. When collected at the end of a camera trapping survey, traditional bait is likely to be ineffective because of decomposition or invertebrate damage, despite being held in a plumbing cowl for protection. In this study I developed two wax block baits aiming to improve bait longevity, whilst not impacting on bait attractiveness. I compare the effectiveness of these two wax block baits against standard peanut butter bait when camera trapping mammals in tropical savanna. I also compared the longevity of the two wax block baits and standard peanut butter bait.

In Chapter five I provide a brief synopsis of the study, including a short discussion of results emerging from the integration of the preceding chapters.

CHAPTER 2: THE BRAMBLE CAY MELOMYS *RUBICOLA* (RODENTIA: MURIDAE): A FIRST MAMMALIAN EXTINCTION CAUSED BY HUMAN-INDUCED CLIMATE CHANGE?

2.1 INTRODUCTION

Anthropogenic climate change is a well-recognised threat to biodiversity globally. This threat is most pronounced in the arctic region, where foraging behaviour and breeding have been altered in many endemic mammal and bird species (Aanes et al. 2000; Stien et al. 2012; Hansen et al. 2013; Hamilton et al. 2015; Descamps et al. 2017). In other parts of the world, abundant evidence exists for range contractions from low latitude and low elevation edges of species' distributions caused by human-induced climate change (Cahill et al. 2012). It is predicted that climate change will contribute to the continued reduction in distribution and potentially cause the extinction of many species over the next 100 years (Galbraith et al. 2002; Baker et al. 2006; Menon et al. 2010; Wetzel et al. 2012; Oliveira et al. 2016).

Australia has some of the highest rates of mammal extinction in the world, with 29 species having become extinct over the past 200 years (Woinarski et al. 2014). Many of these extinctions have been linked to predation by feral cats and foxes, habitat change due to grazing and feral herbivores, habitat loss and fragmentation and the cessation of indigenous burning regimes (Woinarski et al. 2014). The main continental land mass and the large island of Tasmania have borne the brunt of such extinctions. However, five recent Australian mammal extinctions have occurred in species endemic to smaller islands: (1) Lord Howe long-eared bat *Nyctophilus howensis*, (2) dusky or Percy Island flying-fox *Pteropus brunneus*, (3) Christmas Island pipistrelle *Pipistrellus murrayi*, (4) Maclear's rat *Rattus macleari*, and (5) bulldog rat *Rattus nativitatis* (Woinarski et al. 2014). Globally, island mammal species are known to incur very high extinction risks (Medina et al. 2011; Doherty et al. 2016). Scattered around the coastline of Australia are more than 8,000 small islands, and at least 274 of these are found in the Torres Strait (between mainland Australia and Papua New Guinea). Although most of these islands are isolated from the threats listed above, small, low-lying islands are exposed to impacts driven by climate change such as sea-level rise and inundation caused by storm surge from extreme weather events (Department of Climate Change 2009; Green et al. 2010).

Sea-level rise is one of the most certain consequences of global warming. Between 1901 and 2010, global mean sea level rose by 0.19 m and is predicted to rise further by up to 0.82 m by 2100 (IPCC 2014). Sea level will not rise uniformly across geographical regions and therefore some regions may be more susceptible to higher levels of inundation than others (IPCC 2014). On a global scale, ~20% of the world's biodiversity is found on islands and sea-level rise will undoubtedly reduce habitat availability for many species (Bellard et al. 2014). The current impacts of sea-level rise are not well documented for terrestrial fauna as most vulnerable species are subjected to multiple threats. However, impacts of sea-level rise have been linked to a reduction in nesting habitat for marine turtles due to coastal inundation (Fuentes et al. 2010).

Bramble Cay, also known as Maizab Kaur in the eastern Torres Strait language of Meriam Mir, is a tiny, low-lying sand cay located in the north-east Torres Strait that supports the only known population of a murid rodent, the Bramble Cay melomys *rubicola* Thomas, 1924. The species was first documented in 1845 when seamen aboard H.M.S 'Bramble' encountered the cay and observed the animal in high densities (Sweatman 1977). No further accounts of the species were made until December 1978, when as many as several hundred individuals were reported on the cay by Limpus et al. (1983). The first actual assessment of the population was not conducted until 20 years later, when it was estimated that 93 (36 s.e.) individuals were present (Dennis & Storch 1998), making the Bramble Cay melomys one of the world's rarest mammals. Surveys replicating the protocol used by Dennis and Storch (1998) were completed in 2002 and 2004 (P. Latch, pers. comm., 5 June 2015), both indicating a population decline had occurred since the original census (Latch 2008).

The extremely isolated nature of the species' distribution has raised questions about the origin and taxonomic affinities of the Bramble Cay melomys (e.g. Limpus *et al*. 1983; Dennis and Storch 1998; Latch 2008). However, previous studies have confirmed that the species is both morphologically and genetically distinct from other known Australian and New Guinean melomys (Limpus et al. 1983; Dennis & Storch 1998; Bryant et al. 2011).

The Bramble Cay melomys is listed as Critically Endangered under the *IUCN Red List of Threatened Species* (Leary et al. 2008), endangered under the Australian Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC) and endangered under the Queensland *Nature Conservation Act 1992*. In *The Action Plan for Australian Mammals 2012*, Woinarski et al. (2014) recommended changing the EPBC listing to Critically Endangered, given the species is restricted to a small, isolated sand cay and has a small population size that declined at a significant rate between 1978 and 2004. These authors also added '(Possibly Extinct)' to the suggested conservation status based on information communicated to them about the results of preliminary fieldwork from the current study.

The 2008 recovery plan reported that threats to the species included erosion of the cay and introduced plants, predators, competitors and disease, as well as genetic drift and inbreeding depression (Latch 2008). In the face of such threats, both *The Action Plan for Australian Rodents (Lee 1995),* and the national recovery plan (Latch 2008) emphasised the importance of establishing a monitoring program for the Bramble Cay melomys. However, other than the 1998, 2002 and 2004 censuses (Dennis & Storch 1998; Latch 2008), no further monitoring or recovery actions had occurred prior to the present study.

The aim of this work was to confirm the current conservation status of the Bramble Cay melomys and investigate key factors contributing to the species' decline, which involved conducting three surveys on the island between 2011 and 2014. A subsequent aim to salvage any remaining individuals for a captive insurance population was developed following the first two brief surveys.

2.2 MATERIALS AND METHODS

2.2.1 Study area

Bramble Cay is a small (~4 ha), uninhabited, vegetated sand cay located ~50 km from the coast of Papua New Guinea, south-east of the Fly River region (9 08°32.8°°S, 143 52°37.2°°E). It lies within Australian waters, 225 km north-east of the tip of Cape York Peninsula, Queensland (Figure 1). The customary tenure is held by the inhabitants of Darnley Island (Erub), which is located 45 km south–south–west of

Bramble Cay. The cay has a limited history of major human modification, including the establishment in 1924 of a lighthouse, which was demolished in 1954 and eventually replaced by the present lighthouse in 1987.

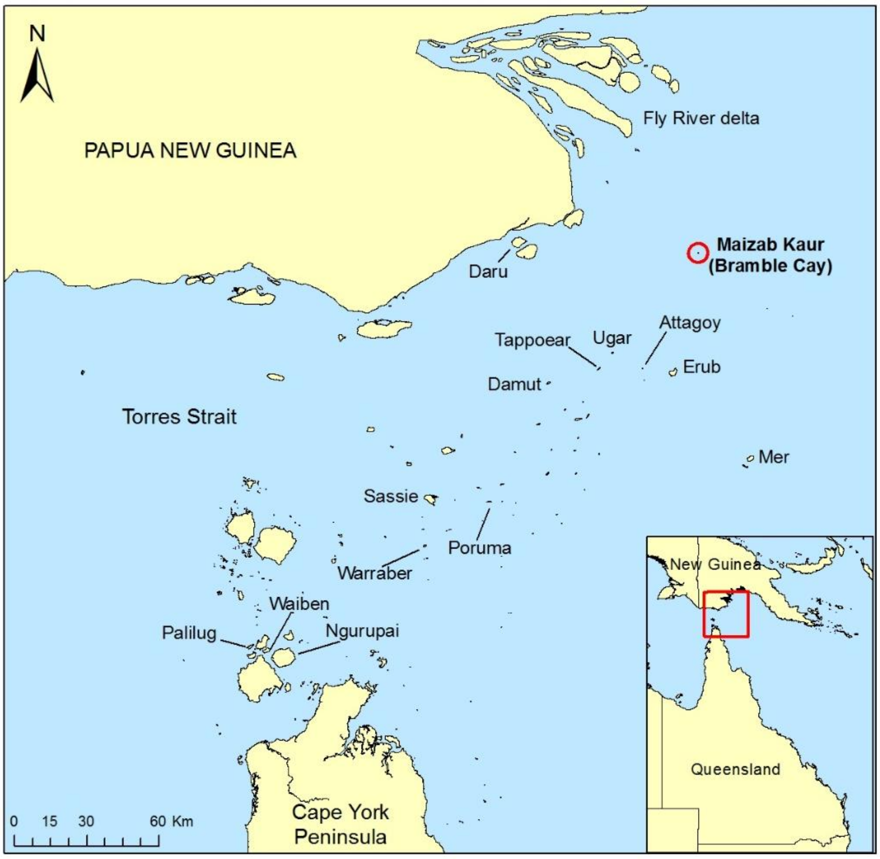

Figure 1. The location of Bramble Cay in relation to Papua New Guinea and Cape York Peninsula, Australia.

At high tide, Bramble Cay has an approximate length of 340 m, a maximum width of 180 m (Dennis & Storch 1998), and a maximum elevation of 3 m (Elvish & Walker 1991). It is formed of diatomaceous sand and surrounded by coral reef (Dennis & Storch 1998). At the south-eastern edge of the cay is a large section of phosphatic rock that rises ~1.9 m above the high tide mark and is in a state of gradual erosion by tides and wave action (Ellison 1998).

Bramble Cay is vegetated with patchy herbaceous cover up to 40 cm in height (Dennis 2012). The composition of the vegetation varies seasonally and/or from year to year, with up to 12 plant (including two weed) species having been recorded (Walker 1988; Latch 2008; Gynther et al. 2014b). Three to five species are typically present at any one time, commonly being *Boerhavia albiflora*, *Portulaca oleracea*, *Cenchrus echinatus* and *Amaranthus viridis* (Dennis 2012). The vegetation is frequently disturbed by the nesting activities of seabirds and green turtles *Chelonia mydas* (Ellison 1998), and unvegetated areas of guano occur where the nesting activities of seabirds are concentrated (Dennis & Storch 1998). The Bramble Cay melomys is the only terrestrial mammal species reported from the island.

2.2.2 Rodent surveys

Bramble Cay was surveyed on three separate occasions: 10–12 December 2011 (by NLW), 17–21 March 2014 (by ICG and ABF) and 29 August–5 September 2014 (by NLW and ICG). Trapping was conducted on each of these visits using a combination of Type A Elliott traps (Elliott Scientific, Upwey, Victoria) and Sherman traps (H.B. Sherman Traps, Tallahassee, FL) in an attempt to confirm the presence of the Bramble Cay melomys and determine a population estimate.

During the December 2011 survey period, transects of 10 traps were placed 5 m apart over two nights. Five such transects were set on the first night and an additional five set the following night, giving a total survey effort of 150 trap-nights. Transects were primarily located within vegetated areas around the perimeter of the cay, where the nesting activity of seabirds is lowest, and along the outcrops of phosphatic rock at the south-eastern end of the island.

Due to the limited extent of vegetation cover during the March 2014 survey, only two transects, each consisting of 20 traps, were set for the three nights, providing an overall effort of 120 trap-nights. One transect was set amidst the most extensive tract of vegetation, as well as targeting beach-washed logs, and the other followed the outcrops of phosphatic rock. During the August–September 2014 survey, 150 traps were deployed for six nights in an approximate grid pattern across sections of the island, thereby achieving a total survey effort of 900 trap-nights. Traps were mostly placed in areas where vegetation was present, with a few also set near or beneath ledges of phosphatic rock.

Trap placement was designed to maximise capture success, as previous observations and surveys indicated the Bramble Cay melomys principally inhabits vegetated areas, avoiding parts of the island supporting the highest densities of nesting seabirds (Limpus et al. 1983; Dennis & Storch 1998). The traps placed along the exposed outcrops of phosphatic rock at the south-eastern end of the island targeted the numerous undercut caves that offered potential daytime refuge for this rodent species. All traps were baited before sunset with standard Australian small mammal bait (Menkhorst & Knight 2011) and checked at sunrise. Due to nesting turtle activity during the December 2011 survey, additional checks were made at 2200 hours and 0200 hours to reduce the chance of any captured animals being killed by turtles crushing or burying traps.

During the August–September 2014 survey, 10 RECONYX Hyperfire HC600 infrared camera traps (RECONYX Inc., Holmen, WI) were also deployed. They were set ~20 m apart around the perimeter of the cay's vegetated area and where nesting birds were less abundant. Camera traps were set at high sensitivity to take three images per trigger event, with no quiet period. Each camera trap was baited with standard Australian small mammal bait (Menkhorst & Knight 2011) inside a stainless-steel tea strainer that was anchored ~1 m from the camera trap. Camera traps were positioned ~35–40 cm above the ground, with the bait located within the centre of the field of view. Baits were changed regularly to ensure freshness.

Nocturnal spotlight and headlamp surveys for the Bramble Cay melomys were completed during the December 2011 and March 2014 surveys. In December 2011, spotlighting was undertaken along each transect during the checking of traps at 2200 hours and 0200 hours using a 300-lm torch (LED LENSER P7, Zweibrüder Optoelectronics GmbH and Co., Solingen, Germany). In March 2014, traps were not checked at night but an active search for the Bramble Cay melomys was conducted across suitable habitat on the island using 250-lm headlamps (LED LENSER H14.2). Total search effort was 2 and 3 person-hours during the December 2011 and March 2014 surveys respectively. Spotlight searches were not undertaken as part of the August–September 2014 survey to ensure any animals present on the island were not disturbed and thus deterred from entering the traps.

Daytime searches of potential shelter sites (e.g. in caves, beneath ledges and under driftwood and other debris) were also conducted for scats, tracks and skeletal material that would indicate the presence or recent occurrence of the Bramble Cay melomys. This effort totalled 5 person-hours of active searching over the three survey periods (3 person-hours in March 2014 and 2 person-hours in August–September 2014, the latter including the use of headlamps to search dark recesses in rocky areas).

Preparations for establishing an insurance population of the Bramble Cay melomys in a captive facility on the Australian mainland are detailed by Gynther et al. (2016).

2.2.3 Cay measurements and environmental data

To document the temporal variation in cay size and vegetation extent, during each of the three visits to Bramble Cay, handheld global positioning system (GPS) devices (a Garmin GPSMAP 60CSx and a GPSMAP 76CSx, Garmin International Inc., Kansas City, MO) were used to record the area of the island above high tide and map the island's vegetated portion. A GPS track along the high-water mark of the highest tide during each survey period was recorded while circumnavigating the island on foot. During the December 2011 survey, GPS tracks were recorded to measure the perimeters of the vegetation. During the two 2014 surveys, GPS tracks were similarly recorded around the perimeters of the major areas of vegetation. However, a measuring tape was used to determine the dimensions of individual scattered patches of vegetation. Both the cay area at high tide and areas of vegetation polygons were calculated using ARCMAP 10.2.2 (ESRI Inc., San Diego, CA).

During the 2014 surveys, the distance between the original lighthouse foundations and the phosphatic rock at the south-eastern end of the cay was determined with a tape measure. Digital photographs were also taken of various aspects of the cay in December 2011 and March and August–September 2014. The Torres Strait Regional Authority (TSRA) supplied photographs from December 2012. Regional rainfall trends were examined using the most reliable long-term data available for the Torres Strait region: yearly rainfall data on Thursday Island (Waiben) and Horn Island (Ngurupai) from the Bureau of Meteorology (2015a).

2.2.4 Information from other sources

We compiled a comprehensive dataset of historical measurements of cay length, width and vegetated extent from published and unpublished sources. We also obtained all available results of past field surveys for the Bramble Cay melomys, and consulted with other researchers and fishermen who may have opportunistically encountered the species when visiting Bramble Cay for other purposes (e.g. turtle monitoring, bird watching, recreation) between 2004 and 2014. Interviews with fishermen also yielded anecdotal information about weather events, sea conditions and floating debris in the vicinity of Bramble Cay.

2.3 RESULTS

2.3.1 Melomys surveys

The survey effort over the three visits to the cay between 2011 and 2014 totalled 1170 small mammal trap-nights, 60 camera trap-nights, 5 h of nocturnal spotlight or headlamp searches and 5 person-hours of diurnal searches for live individuals, tracks, scats and remains of the species. Over the three survey periods, no Bramble Cay melomys were captured in small mammal traps, detected by camera traps or observed during any of the diurnal or nocturnal active searches. Furthermore, no signs of current or former occupation of the cay by the Bramble Cay melomys were found. Capture rates of the Bramble Cay melomys (individuals per 100 trap-nights) declined from ~9.5 in 1998 (Dennis & Storch 1998) to zero in the 2011 and 2014 surveys (Figure 2).

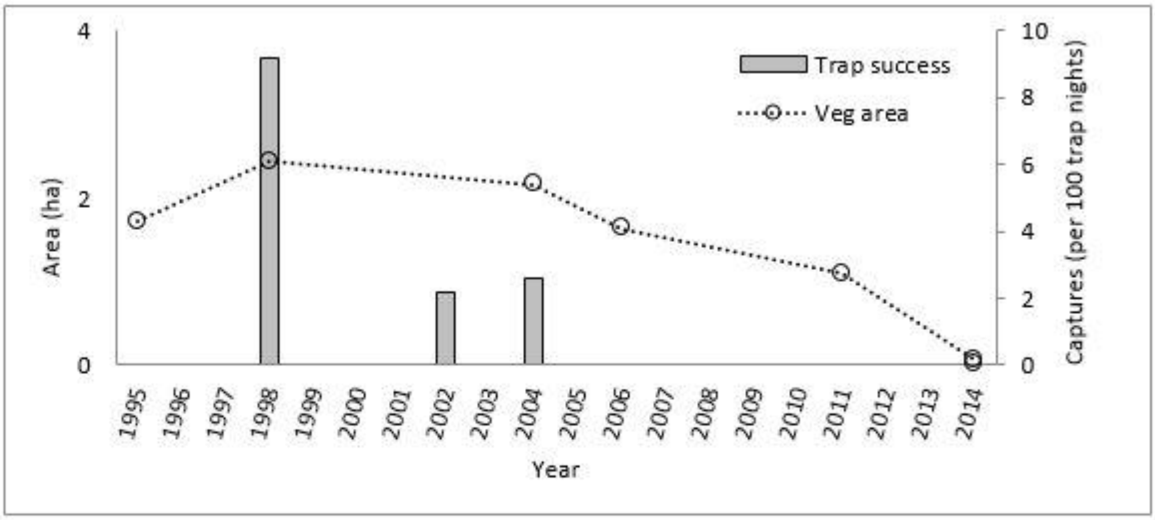

Figure 2. Capture rates of *Melomys rubicola* (individuals per 100 trap-nights) and vegetated area (ha) estimates for Bramble Cay between 1995 and 2014. (Data sourced from Dennis & Storch 1998; Ellison 1998; Latch 2008; P. Latch, pers. comm., 5 June 2015, with 2006 vegetation data calculated from a rectified satellite image).

2.3.2 Cay description

Both the historical data and our own estimates indicated that the area of Bramble Cay has fluctuated over time (Figures 3, 4). The estimated cay area at high tide was 4.55 ha in December 2011, dropping to 2.48 ha in March 2014, before gaining almost one hectare over 5 months to be 3.44 ha in August–September 2014 (Figure. 4). The cay has also experienced significant changes in shape, as is especially apparent by comparing the island’s outline in March 2014 with that in August–September 2014 (Figure 3).

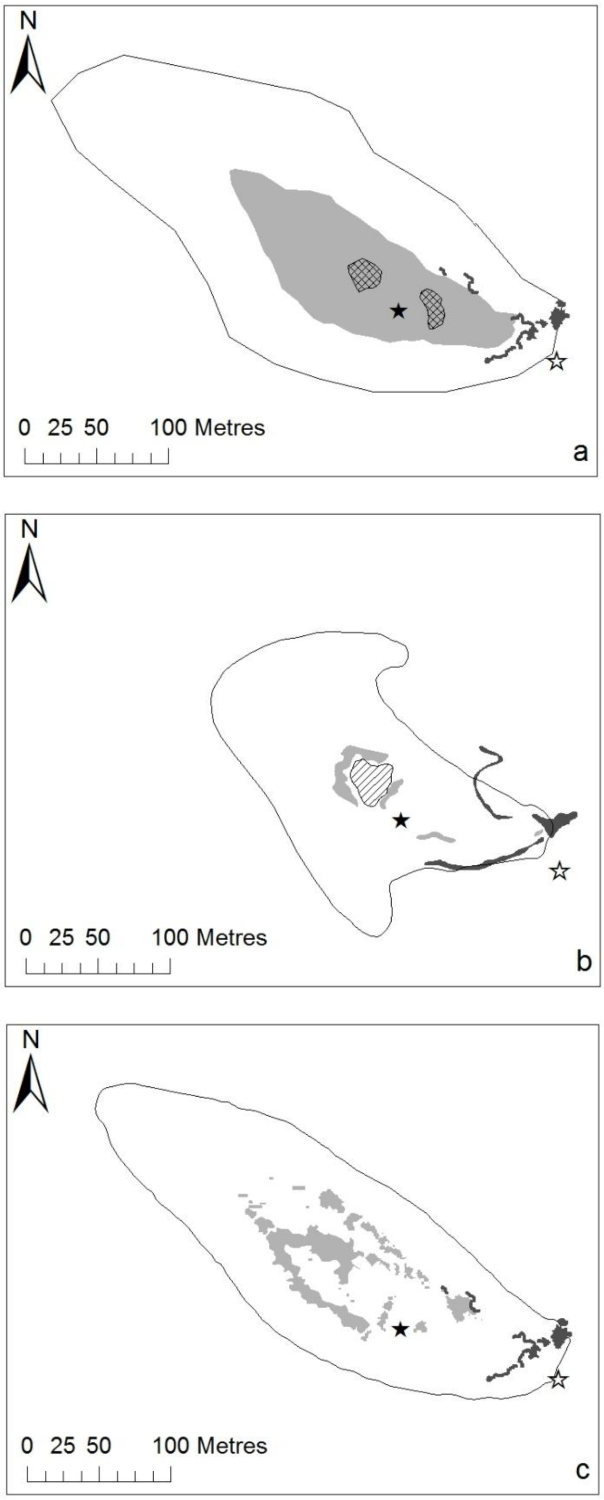

Figure 3. Maps of Bramble Cay illustrating change in vegetation area and cay size between the three survey periods: (*a*) December 2011, (*b*) March 2014 and (*c*) August–September 2014. Key: fine line = cay outline at high tide, dashed line = 1998 vegetation area, dark grey = phosphatic rock outcrops, cross-hatched areas = guano depressions, diagonal striped areas = lagoon, mid-grey = total area of vegetation, black star = current lighthouse, unfilled star = concrete foundations of former temporary lighthouse.

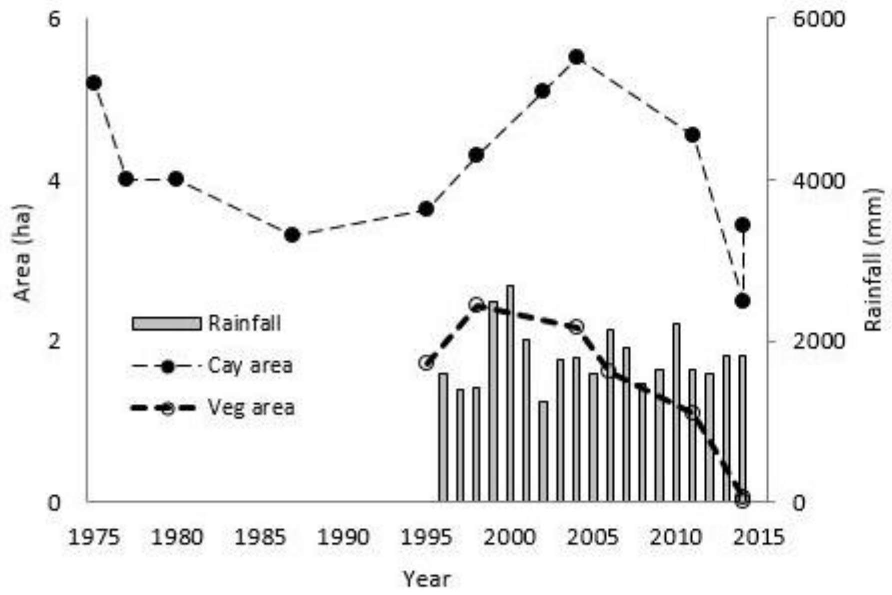

Figure 4. Historical and current data on cay area at high tide and estimated vegetated area (ha) of Bramble Cay, Torres Strait, including rainfall data from Thursday Island and Horn Island, Torres Strait. (Historical data sourced from Limpus *et al*. 1983; Walker 1988; Dennis & Storch 1998; Ellison 1998; Latch 2008).

At the time of Dennis and Storch's (1998) study, the concrete foundations of a 1958 temporary lighthouse tower were situated on top of the phosphatic rock platform and within the vegetated area of the island's south-eastern end. By 2002–2004, they sat at the high tide mark (Latch 2008). By 2011, these foundations were located well down the beach in the intertidal zone (Figure 3). On 4 September 2014, the distance measured between the centre of the concrete foundations and the edge of the phosphatic rock platform at the south-eastern end of the cay was 21 m. Our measurements, therefore, indicate a mean rate of erosion of phosphatic rock from the south-eastern end of Bramble Cay of ~1.3 m year^{-1} over the period of 1998–2014.

The total vegetated area of the cay was 1.1 ha in 2011, but reduced to 0.065 ha in March 2014. Minor recovery occurred over the next 5 months, with the extent of herbaceous cover in August–September 2014 being estimated at 0.19 ha (Figures 2, 3, Appendix 3).

In August–September 2014, clear evidence was collected that showed the cay had undergone at least one recent inundation event. Some vegetation near the southern shoreline that was healthy during the March 2014 visit was dead and uniformly flattened in August–September 2014, suggesting waves had washed over it (Appendix 4a). Furthermore, vegetation located in low-lying areas parallel to the northern shoreline was either dead or badly damaged (Appendix 4b). This apparently salt-affected vegetation occurred in a swale or depression (Appendix 4c), and multiple accumulated masses of driftwood, dead bird eggs and other debris were present at its north-western end (Appendix 4d). The material appeared to have been deposited there via wave action, with the pattern of deposition indicating a flow direction from south-east to north-west.

2.3.3 Anecdotal information

Based on a report from a fisherman who has visited Bramble Cay annually for the 10-year period since 2004, the last known sighting of the Bramble Cay melomys apparently occurred sometime in 2009 (E. Stewart, pers. comm., 4 September 2014; Table 1). Until at least 1983, the species was observed easily by visitors to Bramble Cay (A. Moller-Nielsen, pers. comm., 4 September 2014) and temporarily resident turtle researchers (D. Carter, pers. comm., 5 February 2015), indicating the existence then of a relatively abundant population. The Bramble Cay melomys was not observed by researchers visiting the cay after 2004 (Table 1), suggesting that the species remained at very low abundance for at least 5 years following the last successful survey in 2004 (Latch 2008).

Table 1. Observations of *Melomys rubicola* on Bramble Cay since 2004. Personal communications regarding observations on Bramble Cay since Melomys rubicola was last captured in November 2004. The data in bold indicate the very last sighting of the species.

Time period	**Details**	**Reference**
Nov 2004	Species last captured.	Latch (2008); Peter Latch (pers. comm., 5 June 2015)
Nov/Dec 2007 & 2008	No individuals observed while researchers were camped on the cay.	Dr. Mariana Fuentes (pers. comm., 23 November 2012)
2009	**One or two individuals were disturbed from underneath a pile of sticks and a dug-out canoe when these were flipped over on the north-western beach.**	**Egon Stewart (pers. comm., 4 September 2014)**
Nov/Dec 2009	No individuals observed while researchers were camped on the cay.	Dr. Mariana Fuentes (pers. comm., 23 November 2012)
Dec 2010	No individuals observed when researchers visited the cay.	Karen Evans (pers. comm., 12 December 2011)
Dec 2012	No individuals observed when researchers visited the cay.	Stan Lui (pers. comm., 29 December 2012)
Annual visits during turtle nesting season from 2007	No individuals observed when visiting the cay – deployed 1 baited camera trap at base of lighthouse in 2013 for 3 days, nil individuals photographed.	Shane Preston (pers. comm., 13 March 2015)

Another fisherman (the grandfather of E. Stewart), who has fished the waters around Bramble Cay over a period of 30 years, explained that during high tides of 3.5–4 m, waves break over the south-eastern end of the island (A. Moller-Nielsen, pers. comm., 4 September 2014). He recalled this happening in the late 1980s or early 1990s.

Anecdotes offered by the older of the two fishermen concerning unusual flotsam observed in the sea around Bramble Cay included descriptions of an intact hut adrift on a section of earth and an object that appeared to be a vessel in distress but which, on closer approach, was found to be a floating piece of land that supported an upright palm tree (A. Moller-Nielsen, pers. comm., 4 September 2014). In each case, the flotsam was assumed to have originated from New Guinea, most probably from the Fly River.

2.4 DISCUSSION

2.4.1 The extirpation event

This study confirms the extirpation of the Bramble Cay melomys from Bramble Cay. Our fruitless survey efforts for the Bramble Cay melomys over three separate visits to the island between 2011 and 2014 totalled 1170 small mammal trap-nights, 60 camera trap-nights, 5 h of nocturnal spotlight or headlamp searches and 5 person-hours of diurnal searches for live individuals, tracks, scats and remains of the species. These efforts exceeded the recommended level of sampling for a site of this size (<5 ha) in the *Survey Guidelines for Australia*'*s Threatened Mammals* (DSEWPaC 2011). By comparison, Dennis and Storch (1998) captured 42 Bramble Cay melomys individuals from 444 trap-nights in 1998 (an overall trapping success of 9.5%), and a repeat of this 1998 trap effort in 2002 and 2004 yielded 10 and 12 individuals respectively (Latch 2008; P. Latch, pers. comm., 5 June 2015). Furthermore, the survey conducted by Dennis and Storch (1998) recorded a 21.6% trap success on the first night. Considering these past findings, it is extremely unlikely that the Bramble Cay melomys persists on Bramble Cay but remained undetected during our surveys.

The Bramble Cay melomys appears to have suffered a population decline over the last two or three decades. Although published accounts (e.g. Limpus *et al*. 1983) and unpublished anecdotal reports (this paper) from the late 1970s and early 1980s indicated the species was relatively abundant on Bramble Cay, with a maximum of several hundred individuals present in 1978, Dennis and Storch (1998) estimated the population at 98 individuals in 1998, and Latch (2008) documented a further reduction in capture success in 2002 and 2004 (Figure 2). Because the last known sighting of the species was reportedly made in 2009 (E. Stewart, pers. comm., 4 September 2014), the population on the island was apparently extirpated sometime between then and December 2011.

Our aim to salvage any remaining individuals for a captive insurance population could not be fulfilled due to the loss of this species from Bramble Cay. It appears that efforts to establish a captive insurance population were approximately 5 years too late. The opportunity should have been taken to secure individuals for this purpose sometime between 2002, when surveys first revealed a population decline, and

2009, when the Bramble Cay melomys was last observed. The fate of the Bramble Cay melomys closely follows that of the Christmas Island pipistrelle, which became extinct in 2009, when efforts to capture individuals for an insurance population were made too late (Martin et al. 2012). A recovery plan that provided a framework for research and management of the Bramble Cay melomys was prepared in 2008 (Latch 2008). However, the recovery plan likely came too late in the extinction timeframe and failed to include establishment of a captive breeding program, a management activity required to save the species (Woinarski et al. 2016).

2.4.2 Causes of the extirpation

The reduction in the size of Bramble Cay has been implicated as the main threat to the Bramble Cay melomys (Latch 2008). However, this is not supported by available data (Figures 3, 4), which indicated the cay fluctuates in size seasonally and from year to year. A relatively rapid change in the size of the cay was evident between March and August–September 2014, when the island grew from the lowest documented area of 2.48 ha to 3.44 ha (Figure 4). However, the former measurement coincided with the end of the north-westerly monsoon season, a period when the wind-generated waves are particularly significant for sediment transport on Torres Strait cays and are responsible for shaping the northern shorelines of these reef islands (Parnell & Smithers 2010). The latter measurement was made during the season in which south-easterly trade winds prevail. These seasonally reversing winds undoubtedly also accounted for the dramatic change in shape of Bramble Cay we documented over the interval of just 5 months (Figure 3). These results confirmed that Bramble Cay experiences phases of erosion and deposition, which are likely caused by changes to wind, waves, currents and tides (Duce et al. 2010). Our results and those of other workers (e.g. Limpus *et al*. 1983; Dennis & Storch 1998) do indicate, however, that there is continual erosion of the phosphatic rock from the south-eastern edge of the cay. Although the loss of rock may have reduced sheltering opportunities for the melomys, there is no evidence that this contributed directly to the extinction of the Bramble Cay melomys on the island.

Although little is known of the ecology of the Bramble Cay melomys, this rodent's diet was thought to comprise only or mostly plant material (Limpus et al. 1983; Walker 1988; Dennis & Storch 1998; Ellison 1998; Dickman et al. 2000; Dennis 2012), a diet typical of the *Melomys* genus as a whole (Watts & Aslin 1981). The species primarily used vegetated portions of the cay (particularly those outside of seabird nesting areas) and was frequently observed feeding on *Portulaca oleracea*, a fleshy herb that is common on the island (Dennis & Storch 1998; Latch 2008). The vegetation cover on Bramble Cay has declined since 1998, when ~2.43 ha was present (Figures 2–4). By 2004, the vegetation cover had decreased by 11% to 2.16 ha and declined a further 49% to 1.1 ha by 2011. Our measurements in March 2014 represented the lowest extent of herbaceous cover ever recorded at just 0.065 ha, marking a 94% reduction from 2011 and a 97% loss since 2004, when the Bramble Cay melomys was last trapped during a survey (Latch 2008). The extent of vegetation, however, recovered somewhat to 0.19 ha by the time of the August–September 2014 survey. All but one of these estimates (March 2014) were taken at the end of the dry season and before the onset of the monsoon season, so any seasonal changes in vegetated area should not be reflected in the longer-term trends. The photographs presented in Appendix 3a–d indicate a substantial reduction in vegetation cover occurred between 1980 (before the Bramble Cay melomys declined) and 2012, with further reduction by March 2014. It is perhaps unsurprising then, that the decline in vegetated area coincided with the decline and extirpation of the melomys from Bramble Cay and may be responsible for the species' demise there.

The vegetation on Bramble Cay is regularly disturbed by nesting seabirds (throughout the year) and turtles (from October to March each year) (Walker 1988; Ellison 1998; Latch 2008; Dennis 2012). Damage to the herbaceous cover caused by turtles occurs mainly around the perimeter of the vegetated area (Walker 1988; Latch 2008) and was calculated to represent 20% of the total vegetation cover (2.16 ha) in 2004 (Latch 2008). However, seabirds and turtles are known to have used Bramble Cay as a rookery ever since the island was discovered by Europeans in 1845 (Sweatman 1977), and yet the cay remained well vegetated until at least the early 2000s. Therefore, these nesting activities can be discounted as the cause of the rapid and dramatic reduction in herbaceous cover that occurred post-2004.

Changes in rainfall are not likely to have caused the dramatic reduction in the extent of vegetation on Bramble Cay. Climatic data from Thursday and Horn Islands, ~236 km south-west of Bramble Cay, Coconut Island (Poruma), some 134 km south-west of Bramble Cay (Bureau of Meteorology 2015*a*), and the Papua New Guinean island of Daru, 74 km west of Bramble Cay (Suppiah et al. 2010), indicate that rainfall in the region remained relatively stable during the period over which the vegetation cover declined (Figure 4).

Seawater inundation offers the most plausible explanation for the marked loss of herbaceous cover on Bramble Cay. Both ocean inundation and intrusion of salinity are known to affect terrestrial plants adversely and reduce the resilience of coastal ecosystems (Department of Climate Change 2009; Duce *et al*. 2010). These impacts to low-lying reef islands are anticipated to increase with climate change as a result of rising sea levels, changing weather and oceanographic regimes and a greater frequency and intensity of cyclones (Mimura 1999; Church *et al*. 2006; Mimura *et al*. 2007; Department of Climate Change 2009; Webb and Kench 2010). Globally, the mean sea level has risen by almost 20 cm between 1901 and 2010 (Church et al. 2013), a rate unparalleled in any period during the last 6000 years (Lambeck et al. 2014). The average rate of rise over that time has been 1.7 mm year^{-1}, and ~3.2 mm year^{-1} between 1993 and 2014, with these changes attributable to anthropogenic climate change (Nerem et al. 2010; Church & White 2011; Greenhalgh 2015). Rates of sea-level rise vary regionally (Church et al. 2006; Mimura et al. 2007), with northern Australia experiencing some of the most rapid rates recorded in the world (Church et al. 2010). In the Torres Strait, a tidal gauge at Goods Island (Palilug), ~245 km south-west of Bramble Cay, recorded an average increase in maximum monthly sea levels of ~0.06 cm month^{-1} (equivalent to 7 mm year^{-1}, more than twice the global average) between 1990 and 2014 (PSMSL 2015; Bureau of Meteorology 2015*b*). Furthermore, for the period 1998–2008, anomalies between the measured and predicted tides of as much as 1.15 m were recorded at this tidal gauge, indicating that surge effects can exert more intensive influences on sea levels in the Torres Strait (Duce et al. 2010).

Although cyclones are rare in the Torres Strait because most such systems track further south (Duce et al. 2010; Green et al. 2010), even low intensity, relatively distant cyclones or tropical lows in Australia's Gulf of Carpentaria during the season of prevailing monsoonal north-westerly winds (particularly January and February) can have significant impacts on Torres Strait islands when they occur in conjunction with the high tide (Green et al. 2010; Parnell & Smithers 2010). These systems produce high energy waves during the storm tides, causing severe erosion and inundation (Duce et al. 2010; Green et al. 2010; Suppiah et al. 2010). Such seawater inundation events in January and February 2006, January 2009 and in 2012 have already been documented on as many as seven inhabited islands across the Torres Strait (Duce *et al*. 2010; Green *et al*. 2010; Parnell and Smithers 2010; Department of Environment and Resource Management 2011; Kelly 2014), gaining media attention due to impacts exerted on sea walls and other human infrastructure, as well as culturally important sites (Minchin 2006; Michael 2007; Cohen 2012; Bagnall 2014; Garrett 2014; Kelly 2014). However, even outside of cyclone season, when the Torres Strait experiences a season of reversed wind direction during winter, strong, sustained south-easterlies can produce storm surges and storm tides during the high tide period (Green et al. 2010). An example of this in the northern Torres Strait occurred in July 2005, leading to significant inundation on Murray Island (Mer) and the coastlines of the Gulf and Western Provinces of Papua New Guinea (Green et al. 2010). Other significant flooding and erosion events were noted for inhabited islands of the Torres Strait during 2010 and 2014 (Steffen and Hughes 2013; Torres Strait Regional Authority 2014). The frequency at which inundation events and other impacts associated with extreme sea levels occur will increase with rising mean sea levels, perhaps resulting in permanent inundation of the lowest areas of Torres Strait islands (Department of Climate Change 2009; Duce *et al*. 2010).

The recent increase in cyclonic activity along the east coast of Queensland has been attributed to an alteration in the occurrences of El Niño and La Niña events under the influence of the Inter-decadal Pacific Oscillation (J.J. Callaghan, appendix H in Harper 2013). An analysis of three decades of data from across the entire Pacific Ocean basin determined that occurrences of coastal erosion and flooding are most closely tied to the El Niño/Southern Oscillation. The southern hemisphere, including Torres Strait, experiences more severe conditions during La Niña due to increases in cyclonic activity, wave energy and sea surface elevation (Suppiah et al. 2010; Barnard et al. 2015). Clearly, the damaging impacts exerted on Torres Strait islands, including Bramble Cay, by the changing weather regimes are being driven by climatic oscillations (Barnard et al. 2015). The trend towards a strengthening in the intensity of

La Niña conditions until at least 2012 has been linked to anthropogenic climate change, specifically the increase in global mean temperature (L'Heureux et al. 2013).

Although it is not known whether Bramble Cay has experienced complete inundation during times of peak tides, events involving at least partial inundation have previously been documented. For example, shortly before 1991, saltwater inundation killed the majority of the vegetation on the cay, leaving minimal coverage, although some recovery subsequently occurred (Dennis & Storch 1998). This event very likely corresponds to an anecdotal report of large waves washing over the cay in the late 1980s or early 1990s (A. Moller-Nielsen, pers. comm., 4 September 2014). During the aforementioned July 2005 weather event, very high tides and gale force south-easterly winds that blew for several days reportedly caused waves to be thrown up and over the cay (Latch 2008). This storm may have been responsible for the marked (49%) loss of vegetation cover that occurred on Bramble Cay between November 2004 (Latch 2008) and December 2011 (this study). That an even greater reduction in habitat extent occurred over the ensuing 2 years, reducing the remaining vegetation to just 6% of its 2011 area (Figs 2–4), strongly suggests the island suffered at least one other extreme weather event over that period.

Although *P. oleracea*, the most abundant food resource for the Bramble Cay melomys, is considered salt tolerant, increased exposure to large salt concentrations can cause reduced germination and growth (Yazici et al. 2007). The large areas of dead and damaged *P. oleracea* documented in August–September 2014 (Appendix 4 b & c), along with observed scouring of the substrate overlying rocky areas at the south-eastern end of the cay (plus movement of large logs and other items across the island surface) in the interval between the two visits made during 2014 (Gynther et al. 2016), provided additional evidence of a significant inundation event affecting Bramble Cay over the preceding 5 months. This further demonstrated the island's susceptibility to wave over-wash and seawater intrusion. Such a record of events provides credible evidence that Bramble Cay has experienced ocean inundation on an increasingly regular basis, with significant negative impacts on the extent of vegetation, very likely caused by a greater frequency and/or height of extreme tides and storm surges, coupled with the rising sea level, resulting from anthropogenic climate change. The likelihood that the island would be subjected to such impacts driven by climate change was predicted. For example, Latch (2008) suggested that because Bramble Cay is low-lying and sits exposed on a shallow reef flat, it is vulnerable to the effects of climate change, resulting in stronger winds and larger storm surges. Turner and Batianoff (2007) warned that a severe cyclone may destroy all of the island's vegetation, while Woinarski et al. (2014) suggested rising sea levels may inundate the island and, consequently, the whole range of the Bramble Cay melomys.

Over and above the deleterious effects on the Bramble Cay melomys caused by the loss and degradation of habitat resulting from such repeated inundation, Woinarski et al. (2014) recognised that wave action associated with severe storms may have direct and catastrophic impacts on the animals themselves, with the potential to destroy this rodent's entire population. It seems probable that a combination of these threats, worsening over the last two decades, finally extirpated the Bramble Cay melomys on Bramble Cay (Gynther et al. 2016; Watson 2016). The loss of this unique population may represent the first documented mammal extinction due solely (or primarily) to anthropogenic climate change. The *IUCN Red List of Threatened Species* indicates that climate change and severe weather played a role in one other mammalian extinction in the 1950s, that of the Little Swan Island hutia *Geocapromys thoracatus*. However, even though a single, very strong hurricane in 1955 may have been a factor in the species' decline, the main threat to this island population was actually predation by introduced cats (Turvey & Helgen 2008).

Although the loss of the Bramble Cay melomys is thought to be the first mammal extinction brought about by anthropogenic climate change, it is unlikely to be the last. Numerous studies have investigated the potential future impacts of sea-level rise on terrestrial fauna (Galbraith et al. 2002; Baker et al. 2006; Menon et al. 2010; Wetzel et al. 2012; Bellard et al. 2014; Oliveira et al. 2016). Future extinctions due to sea-level rise are predicted to be most prominent in north-eastern South America, although smaller numbers of extinctions are predicted to be scattered worldwide (Menon et al. 2010). It is estimated that a sea-level rise of 1 m will cause a 0.7% loss of total global land area (Menon et al. 2010), completely submerge 6% of islands in 10 biodiversity hotspots across the world (Bellard et al. 2014) and inundate 3% of all coastal areas in the South-east Asian and Pacific region (SEAP) (Wetzel et al. 2012). Wetzel et al. (2012) predict 11–15% of potential habitat on islands will be lost in the SEAP region due to inundation and erosion, with some of this loss being attributed to human refugee migration inland.

The demise of the Bramble Cay melomys and the future predictions of habitat losses worldwide due to ocean inundation highlight the immediate need to consider the impacts of sea-level rise and other adverse oceanographic impacts stemming from human-induced climate change in conservation planning. Prioritisation of conservation actions on islands and coastal areas that provide habitat for vulnerable species threatened by ocean inundation may assist in reducing future biodiversity losses.

2.4.3 Extralimital populations

In light of the extirpation of the Bramble Cay melomys on Bramble Cay, hopes for the species' persistence rest upon locating other, as yet unknown, populations. Given that the Bramble Cay melomys was first documented 170 years ago (Sweatman 1977) and formally described some 90 years ago (Thomas 1924), but has never been found anywhere else in the meantime, the likelihood that additional populations exist appears to be low. However, Latch (2008) suggested that other Torres Strait islands and the Fly River region of New Guinea are locations that may support the Bramble Cay melomys.

Fauna surveys conducted on the 17 inhabited islands in the Torres Strait are summarised by Lavery et al. (2012). The three inhabited islands closest to Bramble Cay, i.e. Stephen's Island (Ugar; 9 50°70°°S, 143 54°50°°E), Murray Island (9 91°67°°S, 144 05°00°°E) and Darnley Island (9 5°870°°S, 143 77°10°°E), all support the grassland melomys *M. burtoni*, but not the Bramble Cay melomys (Lavery et al. 2012). Recent surveys conducted in March 2014 on three uninhabited coral cays within close proximity to Bramble Cay also failed to detect the species (Fell & Gynther 2014a, 2014b; Fell & Gynther 2014c; Gynther et al. 2014b). In light of these findings and due to the lack of records of the Bramble Cay melomys from museum collections (Queensland Museum, Brisbane; Australian Museum, Sydney) and other extensive fauna survey work conducted across islands in Torres Strait (e.g. Limpus *et al*. 1983; Lee 1995; Ingram and Caneris 2004*a*, 2004*b*; Ingram 2008; Watson 2009; Diete 2010; Hitchcock 2012; Watson 2012; Fell and Watson 2014; Gynther *et al*. 2014a, Gynther *et al.* 2014b; Reis *et al*. 2015) it is unlikely that the Bramble Cay melomys remains in the Torres Strait.

A species is eligible to be considered extinct in the wild under Queensland's *Nature Conservation Act 1992* and the federal EPBC and, internationally, to be presumed

Extinct under the IUCN Red List (IUCN 2001) if it has not been recorded in its known and/or expected habitat, at appropriate seasons, anywhere in its past range, despite exhaustive surveys over a time frame appropriate to its life cycle and form. In the case of the Bramble Cay melomys, the recent surveys of Bramble Cay were both 'thorough' and 'exhaustive' and sampled all 'known habitat' at appropriate times, and represented the full extent of the known historic range of the species. Furthermore, the Bramble Cay melomys is unlikely to be found on other islands in the Torres Strait. Consequently, it is reasonable to conclude that the species is extinct in the wild in Queensland and Australia, and extinct globally.

Based upon a lower genetic affinity to known melomys species from New Guinea and the closer relationship between the Bramble Cay melomys and Cape York melomys *M. capensis* in Australia, the presence of the Bramble Cay melomys on Bramble Cay is possibly a result of it having been stranded by the sea-level rise that followed the Last Glacial Maximum flooding the land bridge between Australia and Papua New Guinea (Dennis & Storch 1998; Bryant et al. 2011). However, other speculation suggests that the Bramble Cay melomys population on Bramble Cay is derived from a population somewhere in the Fly River region of New Guinea (Dennis & Storch 1998; Latch 2008). Bramble Cay is located only some 50 km from the mouth of the Fly River, and logs and debris are regularly deposited on the shoreline of the cay, very likely transported by the river when in flood (Dennis & Storch 1998; Gynther et al. 2014b; Gynther et al. 2016). Large amounts of debris, including a floating island (6 m 6 m) supporting an upright palm tree and, in one case, a small human dwelling (a hut), have been seen at sea in the vicinity of Bramble Cay, presumably originating from the Fly River (A. Moller-Nielsen, pers. comm., 4 September 2014; D. Carter, pers. comm., 5 February 2015). Although the Bramble Cay melomys has not been identified among Papua New Guinean fauna (Waithman 1979; Flannery 1995; Menzies 1996), mammal records from the Fly River region are considerably lacking (Lavery et al. 2013) and it remains a very small possibility that the Bramble Cay melomys exists there (Bryant et al. 2011; Dennis 2012). Therefore, our best hope for this species is that an undiscovered population occurs in the Fly River delta.

2.5 CONTRIBUTION TO AUTHORSHIP

The following outlines my contribution to the authorship of this publication and chapter in the thesis:

I made substantial contribution to the conception and design of the project. I originally visited Bramble Cay in 2011 as the sole ecologist working on the species. I undertook the first survey independently with in-kind support from the Torres Strait Regional Authority. I also visited Bramble Cay in 2014 and worked in collaboration with the Department of Environment and Heritage Protection to design and implement the survey aimed to recover the species. On both field surveys I organised a substantial amount of the field equipment and other logistics. I set and collected all camera traps on the 2014 survey and reviewed all data. During both surveys I collected field data and was involved in the planning of trap placement.

I collated the historical data, cay measurements and other environmental data and created the maps and figures used in the publication, with exception of Figure 1 (location of Bramble Cay) which was created by the Department of Environment and Heritage Protection. I interpreted a large proportion of the research data with some assistance from Dr. Ian Gynther and Dr. Tyrone Lavery. I wrote the majority of the discussion which was then edited by my supervisor Dr. Luke Leung and co-author Dr. Ian Gynther.

CHAPTER 3: LOW CAPTURE SUCCESS OF SMALL AND MEDIUM SIZED MAMMALS: DOES CAMERA TRAPPING INCREASE THE DETECTABILITY OF MAMMALS IN NORTHERN QUEENSLAND?

3.1 INTRODUCTION

Since European settlement, Australia has experienced an alarming rate of decline in native mammalian fauna, with 29 endemic land mammals becoming extinct (McKenzie et al. 2007; Burbidge et al. 2008; Woinarski et al. 2015; Waller et al. 2017). There is clear evidence that recently, regional extinctions and reductions in range distributions (some as much as 90%) of small and medium sized mammals have occurred in parts of tropical northern Australia (Braithwaite & Griffiths 1994; Fitzsimons et al. 2010; Woinarski et al. 2010; Woinarski et al. 2011a; Frank et al. 2014; Wayne et al. 2017).

A number of factors have been implicated for the decline of northern Australian mammals, including altered fire regime, grazing of introduced herbivores, drought, predation by cats and wild dogs, impacts of introduced cane toads and exotic wildlife disease (Woinarski et al. 2011a; Kutt 2012; Fisher et al. 2013; Reiss et al. 2013; Frank et al. 2014). Identifying the principle cause, however, is difficult because historical declines occurred simultaneously with many changes to Australian ecosystems and current land use patterns and fire varies geographically in scale and intensity (Burbidge et al. 2008). Many of these mammals persist in critically low densities that experimentally evaluating factors influencing populations is not logistically possible.

Although surveys undertaken in other regions of northern Australia have documented declines in small and medium sized mammals (Woinarski et al. 2004; Woinarski et al. 2010), it is unknown if these declines have extended to the Northern Gulf Management Region, Queensland. Limited fauna surveys have been undertaken in the Northern Gulf Management Region due to its remoteness. Surveys are often expensive, labour intensive or located in areas that are difficult to access, and very few study sites have been re-surveyed. Fauna survey data for the Northern Gulf Management Region, typically identifies a low abundance and species diversity of small and medium sized mammals. The low capture success of mammals and lack of re-surveys, therefore, limits the ability of researchers to determine if these populations have declined. Identifying methods to improve the detection of small and medium sized mammals is crucial to establishing monitoring programs for potentially declining mammal populations in the region.

Camera traps are increasingly being used in Australia to survey fauna populations as they offer many advantages over traditional fauna survey techniques that rely on live-capture of animals (Burton et al. 2015; Meek et al. 2015a). Camera traps can be deployed for extended time periods (up to several months) without intervention (McDonald et al. 2015), making this method particularly useful for remote areas (Gillespie et al. 2015). Camera trapping may increase the detection probability of rare or cryptic species that are difficult to capture in live traps (McDonald et al. 2015). Several studies have proven the effectiveness of camera trapping for detecting small and medium sized mammals in Australia, particularly when camera traps are deployed for longer periods (Claridge et al. 2010; De Bondi et al. 2010; Paull et al. 2011; Paull et al. 2012; Welbourne et al. 2015). However, only two of these studies compared camera trapping and live trapping methodologies for detecting small Australian mammals (De Bondi et al. 2010; Welbourne et al. 2015). De Bondi et al. (2010) found that camera trapping yielded comparable results as live trapping but was considerably more cost effective, which could be useful for increasing sample size. Welbourne et al. (2015) however, found that camera traps detected significantly more small mammal species per transect than the other survey methodologies tested (a mixture of live traps and refuge arrays). Although the initial cost of camera traps was comparatively high, camera traps were considered more cost effective in the long-term (Welbourne et al. 2015). Thus, camera traps may be the key to improving the detection of small and medium sized mammals in vast landscapes of north Queensland. To date no studies have been previously conducted to experimentally compare detection rates of camera traps versus live trapping protocols in north Queensland.

This study provides a collation of mammal population survey data collected in the Northern Gulf Management region, for species up to approximately five kilograms in body weight. We report on the relative abundance of mammals across the region and compare capture rates between bioregions (Einasleigh Uplands versus Gulf Plains) within the region and between this region and other regions of Northern Australia where declines in some small and medium sized mammal species have been evident. We integrated camera traps into the systematic plot-based survey method, which is widely used across northern Australia, to critically determine if camera trapping improved the detection of small and medium sized mammals in northern Queensland when integrated into this method.

3.2 METHODS

3.2.1 Study sites

The Northern Gulf Management Region, Queensland, encompasses approximately 196,921 km^2 of Northern Queensland and is dominated by two bioregions, the Gulf Plains and Einasleigh Uplands. The Gulf Plains bioregion is characterised by its vast alluvial flats with sandstone outcrops along its eastern margin with elevations up to 1000 m above sea level. The neighbouring Einasleigh Uplands bioregion straddles the Great Dividing Range; with varying altitudes from 100 m to 1100 m. Both bioregions are dominated by open tropical savanna woodlands. Approximately 90% of the region is under pastoral lease for grazing, with secondary land uses such as cropping and mining occurring in some areas.

A total of 276 1-hectare sample plots on seventeen properties across the Northern Gulf Management Region (Figure 5) were surveyed for terrestrial fauna between 2003 and 2016 without the use of camera traps. Of these, 149 sample plots were surveyed once, 100 sample plots twice and 27 sample plots three times, giving a total of 430 plot surveys. Of these, 69 sample plots were located in the Gulf Plains bioregion (108 plot surveys) and 207 in the Einasleigh Uplands bioregion (322 plot surveys). Surveys were conducted by the candidate (in 2013 to 2016), staff and consultants engaged by Northern Gulf Resource Management Group (prior to these dates).

A further 63 1-hectare experimental plots were surveyed once during 2015 and 2016 which integrated camera traps into the methodology. Of these, 52 were located in the Cape York Management Region and 11 in the Northern Gulf Management Region (Figure 5).

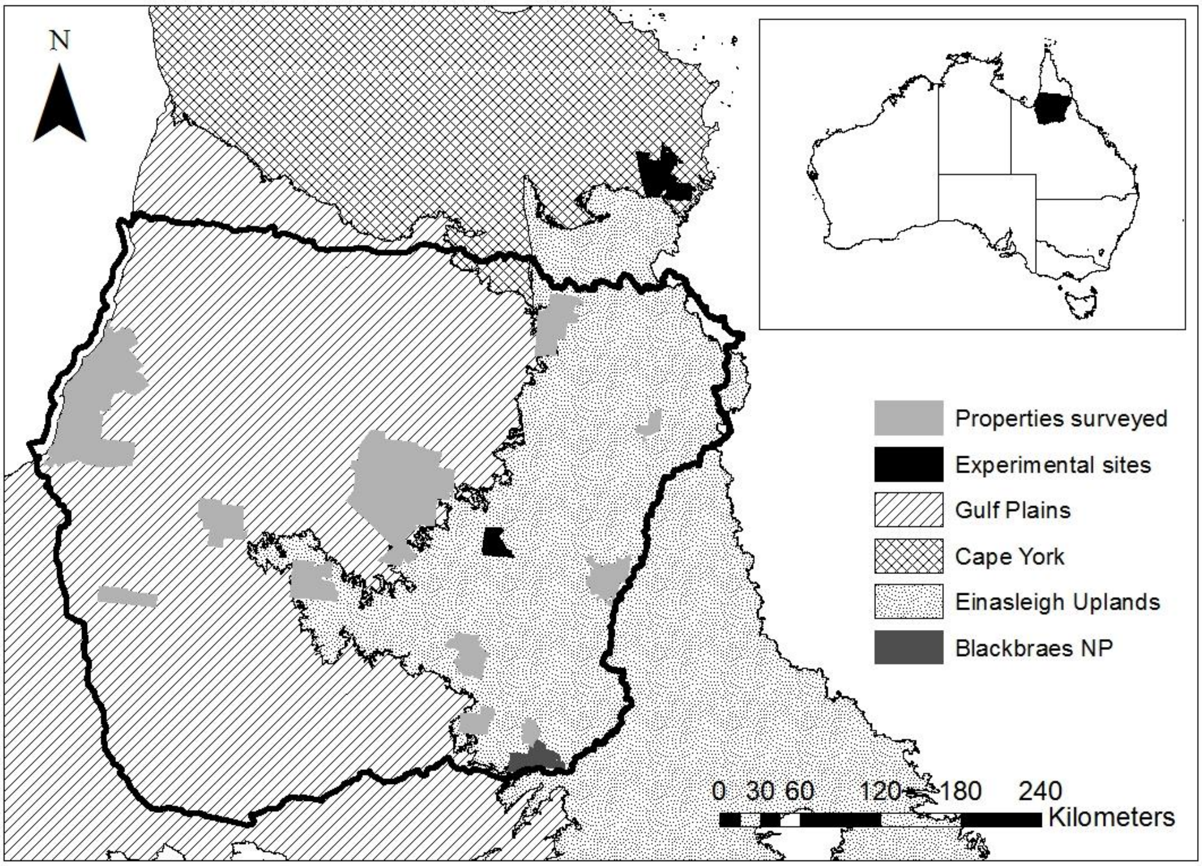

Figure 5. Map showing the location of surveyed properties, experimental sites using camera traps, the three bioregions sampled, and extent of Northern Gulf Management Region (black line). Blackbraes National Park in the south-eastern corner was the only protected area included in the study.

3.2.2 Standard plot survey methods

Fauna sampling methods used in this study broadly followed the recommendations of the 'Terrestrial Vertebrate Fauna Survey Guidelines for Queensland' based on the 1 ha sampling plot methodologies by Eyre et al. (2012). The plot consisted of 20 'Type A' (330 x 100 x 90 mm) Elliot traps placed 10 m apart (either in a 50 x 50 m square or along 2 traplines 50m apart), four pitfall traps (either in a T shaped configuration or spaced evenly in the plot), and two cage traps placed at either end of the plot. Some plots were only partially sampled and therefore reflected in the trapping effort.

Plots were placed at least 1 km apart for sampling independence. Each plot was trapped for three or four consecutive nights at each sampling session. Elliot traps were baited with a mixture of flour or oats, vanilla essence, honey, peanut butter, and dog biscuits or sardines. Cage traps were baited with jam sandwiches and luncheon meat. Newly captured individuals were given a unique marking for individual identification. Spotlighting occurred on each plot twice for a total of 20-30 minutes in each plot. Animals sighted in a spotlight survey were likely to be different individuals as repeated counting of an individual was avoided. Mammals recorded in the study were up to 5 kg in body weight, including the common brushtail possum and short-beaked echidna. Bats were not targeted by this sampling design.

3.2.3 Camera trapping survey methods

Two RECONYX Hyperfire camera traps were integrated into each of the 63, 1-hectare experimental plots surveyed during 2015 and 2016. Camera traps were placed 50 m apart along the centre line of each plot. Camera traps were secured on a tree 30 cm above the ground and directed at a point 1 to 2 metres away. A steel post would be used to secure the camera trap if a tree was not available. The camera trap was baited with a mixture of peanut butter, honey, vanilla essence and oats. Bait was secured in a 50 mm sewer PVC ventilation cowl placed 1 to 2 metres from the camera. Vegetation was cut to ground level in the area between the camera trap and cowl to mitigate the camera trap misfiring. Camera traps were deployed for thirty-five days (± 2 days) prior to trapping and spotlighting on the plot. A mix of white flash (HC550) and infrared flash (HC600) types were used, depending on availability.

3.2.4 Species Identification

Camera trap photographs were identified to species or genus where possible. Photographs of animals where clear identification to species or genus was not possible, identification was made to family or order. Trapping data, distribution and physical features (e.g. coat colour, tail length and size of the animal) were used to assist with identification of rodent species. To distinguish between *M.burtoni* and *M. cervinipes*, habitat type and trapping data was used to determine identification. *P. delicatulus* and *P. gracilicaudatus* were identified separately using size and coat colour (e.g. *P. delicatulus* is much smaller than *P. gracilicaudatus* and has distinctive fur colouration).

3.2.5 Event definitions

The relative abundance of species photographed was quantified by the number of photographic events per sampling session. An event is defined as one or a sequence of consecutive photographs of an individual of no more than a species-specific interval apart. The probability of an event caused by the recapture of an individual should be equal to or no greater than that caused by a newly captured individual. This species-specific interval was determined with photographic data of known individuals identified by natural marking to be 15 minutes for the northern quoll and northern brown bandicoot, and 30 minutes for the agile wallaby by Diete et al. (2015) and the common brushtail possum as per Chapter 4. For all the other study species 60 minutes was set as the interval for event definition due to the low number of occurrences.

The number of days until the first event for each species on each plot was calculated by comparing the date and time of the first photographic event to the date and time of camera deployment.

3.2.6 Statistical analysis

A modified McNemar test was used to test the disagreement in the detection of small and medium sized native mammals and introduced species, dingo, pig and cat, between traditional plot survey methods and camera trapping across the experimental plots. Species that were detected on less than five plots were excluded from the analysis due to small sample size.

3.3 RESULTS

3.3.1 Plot surveys

We combined datasets from 430 1-ha plot surveys, which represented sampling efforts totalling 3,316 cage trap nights; 34,480 Elliot trap nights; 6,270 pitfall trap nights and 298.5 hours of spotlighting. A total of 461 individuals, from 24 species were captured from these methods (Table 2; Table 3). The overall trap success of small and medium sized mammals varied between capture methods. Cage traps yielded the highest trap success (3.79 captures per 100 trap nights), followed by pitfall traps (0.51 captures per 100 trap nights) and Elliott traps (0.35 captures per 100 trap nights). Spotlighting yielded 0.59 captures (sighting of an animal) per 60 minutes.

Of the 430 plots, 261 failed to detect any small or medium sized mammals, particularly in the Gulf Plains bioregion (Table 3). The majority of small and medium sized mammal records in the Einasleigh Uplands bioregion were restricted to Blackbraes National Park and its neighbouring property (Werrington Station), where 106 of 131 (80.9%) of the plots recorded small and medium sized mammals. Accumulated trap success for Blackbraes National Park and Werrington Station were 11.5, 0.7 and 1.0 captures per 100 trap nights for cage, Elliott and pitfall traps respectively and 1.52 records per hour of spotlighting. In comparison, trap success for the remainder of the Northern Gulf region were 0.3, 0.2 and 0.3 captured per 100 trap nights for cage, Elliott and pitfall traps respectively and 0.17 records per 100 hours of spotlighting.

Table 2. The mean number of species captured per plot survey from the 430 plot surveys (2003-2016). Recording success is calculated as individuals per 100 trap nights, or animals sighted per hour of spotlighting.

		Method					Recording success		
Common name	**Scientific name**	**C**	**E**	**P**	**S**	**Total**	**Trapping**	**Spotlighting**	**Mean**
Small sized mammals									
Rodentia									
Muridae									
Lakeland Downs mouse	*Leggadina lakedownensis*		16	8		**24**	0.0589	-	0.0557
Black-footed tree-rat	*Mesembriomys gouldii*	1				**1**	0.0301	-	0.0023
House mouse	*Mus musculus*		39	2		**41**	0.1006	-	0.0951
Delicate mouse	*Pseudomys delicatulus*		1	1	1	**3**	0.0049	0.0100	0.0070
Eastern chestnut mouse	*Pseudomys gracilicaudatus*		35		1	**36**	0.1015	0.0033	0.0835
Eastern pebble-mound mouse	*Pseudomys patrius*		13	1	1	**15**	0.0343	0.0033	0.0348
Black rat	*Rattus rattus*	1			1	**2**	0.0301	0.0033	0.0046
Long-haired rat	*Rattus villosissimus*		2			**2**	0.0058	-	0.0046
Pale field-rat	*Rattus tunneyi*	1	14	1		**16**	0.0363	-	0.0371
Marsupiala									
Dasyuridae									
Common planigale	*Planigale maculata*		1	12		**13**	0.0319	-	0.0302
Long-tailed planigale	*Planigale ingrami*			6		**6**	0.0956	-	0.0139
Chestnut dunnart	*Sminthopsis archeri*			1		**1**	0.0159	-	0.0023
Common dunnart	*Sminthopsis murina*		1		1	**2**	0.0029	0.0033	0.0046
Petauridae									
Sugar glider	*Petaurus breviceps*				20	**20**	-	0.0670	0.0464
Squirrel glider	*Petaurus norfolcensis*				9	**9**	-	0.0301	0.0209
Petaurus species	*Petaurus spp.*				1	**1**	-	0.0033	0.0023
	Total	3	122	32	35	**192**			
Medium sized mammals									
Lagomorpha									
Leporidae									
European rabbit	*Oryctolagus cuniculus*		1		14	**15**	0.0290	0.0469	0.0348
Monotremata									
Tachyglossidae									
Short-beaked echidna	*Tachyglossus aculeatus*				3	**3**	-	0.0100	0.0070
Marsupialia									
Peramelidae									
Northern brown bandicoot	*Isoodon macrourus*	5	1		1	**7**	0.0159	0.0033	0.0162
Macropodidae									
Rufous bettong	*Aepyprymnus rufescens*	38			36	**74**	1.1459	0.1206	0.1717
Spectacled hare-wallaby	*Lagorchestes conspicillatus*	2			5	**7**	0.0603	0.0167	0.0162
Allied rock-wallaby	*Petrogale assimilis*				1	**1**	-	0.0033	0.0023
Mareeba rock-wallaby	*Petrogale mareeba*	1				**1**	0.0301	-	0.0023
Phalangeridae									
Common brushtail possum	*Trichosurus vulpecula*	77			29	**106**	2.3220	0.0971	0.2459
Pseudocheiridae									
Greater glider	*Petauroides volans*				55	**55**	-	0.1842	0.1276
	Total	123	2	0	144	**269**			
	Grand Total	**126**	**124**	**32**	**179**	**461**			

Key: C=cage, E=Elliott, P=pitfall, S=spotlighting.

Table 3. Summary table of survey effort, abundance and diversity of small and medium sized mammals for each of the bioregions (430 plot surveys 2003-2016). The relative abundance of species was the number of animals caught per 100 trap-nights (cage, Elliott and pitfalls). The relative abundance for spotlighting data was the number of animals encountered per hour.

	Einasleigh Uplands	**Gulf Plains**
Sample plots surveyed (*n*)	322	108
Plots (*n*) with nil < 5kg mammal captures	169 (52.48%)	92 (85.18%)
Total no. species recorded	23	9
Average species richness per plot	0.85	0.15
Relative abundance (trapping)	0.81	0.12
Relative abundance (spotlighting)	1.31	0.07

3.3.2 Experimental plots

Twenty-five small and medium sized mammal species were detected during the experimental plot surveys in 2015-2016, comprising 20 native and 5 introduced species (including the dingo) (Table 4). There were 44 events that were rodents, but we were not able to identify them to genus or species level. Of the 539 captures that could be confidently identified, 82.0% (442 camera events from 20 species) were detected by camera trapping, and 18.0% (97 captures from 15 species) were captured using the standardised plot survey method. Ten species (both native and introduced) were detected exclusively by camera traps (5 small and 5 medium sized mammals) and five by the standard plot survey method (5 small mammals).

There were significant disagreements in the detection of small and medium sized mammals between traditional plot survey methods and camera traps (Table 5). Camera traps were significantly more effective than the standard plot survey in detecting the dingo, pig, northern quoll, northern brown bandicoot and short-beaked echidna. The grassland melomys was detected on significantly more plots using the traditional survey plot method. Eight native species were detected exclusively by camera traps and four native species were detected exclusively by the standard plot survey methodologies.

Table 4. **The total and mean number of each species captured per plot from the 63 experimental plots (2015-2016).**

Common name	Scientific name	Method C	E	P	S	Cam	Total	Mean
Small sized mammals								
Rodentia								
Muridae								
Black rat	*Rattus rattus*					2	**2**	0.0317
Black-footed tree-rat	*Mesembriomys gouldii*					3	**3**	0.0476
Canefield rat	*Rattus sordidus*		5			40	**45**	0.7143
Common rock rat	*Zyzomys argurus*		2			22	**24**	0.3810
Delicate mouse	*Pseudomys delicatulus*		1	5		3	**9**	0.1428
Eastern chestnut mouse	*Pseudomys gracilicaudatus*		1				**1**	0.0159
Fawn-footed melomys	*Melomys cervinipes*		47	1	3	45	**96**	1.5238
Giant white-tailed rat	*Uromys caudimaculatus*	1				5	**6**	0.0952
Grassland melomys	*Melomys burtoni*		10	1	1	1	**13**	0.2063
House mouse	*Mus domesticus*		1				**1**	0.0159
Lakeland downs mouse	*Leggadina lakedownensis*			1			**1**	0.0159
Rakali water rat	*Hydromys chrysogaster*					1	**1**	0.0159
Unidentified rodent species	Rodentia sp.					44	**44**	0.6983
Marsupialia								
Dasyuridae								
Brush-tailed phascogale	*Phascogale tapoatafa*					3	**3**	0.0476
Common planigale	*Planigale maculata*			1			**1**	0.0159
Northern quoll	*Dasyurus hallucatus*	1	1			50	**52**	0.8254
Red-cheeked dunnart	*Sminthopsis virginiae*					3	**3**	0.0476
Petauridae								
Sugar glider	*Petaurus breviceps*				5		**5**	0.0794
Sub total		**2**	**68**	**9**	**9**	**222**	**310**	
Medium sized mammals								
Carnivora								
Felidae								
Cat	*Felis catus*				2	7	**9**	0.1429
Canidae								
Dingo	*Canis lupus dingo*				1	58	**59**	0.9365
Marsupialia								
Phalangeridae								
Common brushtail possum	*Trichosurus vulpecula*	1	1		4	26	**32**	0.5079
Macropodidae								
Godman's rock wallaby	*Petrogale godmani*					1	**1**	0.0159
Rufous bettong	*Aepyprymnus rufescens*					1	**1**	0.0159
Peramelidae								
Northern brown bandicoot	*Isoodon macrourus*					47	**47**	0.7460
Suidae								
Suinae								
Pig	*Sus scrofa*					64	**64**	1.0159
Monotremata								
Tachyglossidae								
Short-beaked echidna	*Tachyglossus aculeatus*					16	**16**	0.2540
Sub total		**1**	**1**	**0**	**7**	**220**	**229**	
Grand Total		**3**	**69**	**9**	**16**	**442**	**539**	

Key: C=cage, E=Elliott, P=pitfall, S=spotlighting, Cam=camera.

Table 5. The number of experimental plots where animals were detected using traditional survey techniques (T) and camera traps (C). The total is the total number of experimental plots where animals were detected, including plots where species were detected using both methods.

		Number of plots				
Common name	**Scientific name**	**T**	**C**	**Total**	**√ mcnema statistic**	**p-value**
Small sized mammals						
Rodentia						
Muridae						
Black-footed tree-rat	*Mesembriomys gouldii*	0	1	1	1.00	-
Black rat	*Rattus rattus*	0	1	1	1.00	-
Canefield rat	*Rattus sordidus*	2	2	2	0.00	-
Common rock rat	*Zyzomys argurus*	2	3	3	1.00	-
Delicate mouse	*Pseudomys delicatulus*	5	2	5	-1.73	-
Eastern chestnut mouse	*Pseudomys gracilicaudatus*	1	0	1	-1.00	-
Fawn-footed melomys	*Melomys cervinipes*	9	8	9	-1.00	*0.2500*
Giant white-tailed rat	*Uromys caudimaculatus*	1	2	3	0.58	-
Grassland melomys	*Melomys burtoni*	7	1	8	-2.12	*0.0176*
House mouse	*Mus domesticus*	1	0	1	-1.00	-
Lakeland downs mouse	*Leggadina lakedownensis*	1	0	1	-1.00	-
Rakali water rat	*Hydromys chrysogaster*	0	1	1	1.00	-
Unidentified rodent species	*Rodentia sp.*	0	16	16	4.00	*<0.001*
Marsupialia						
Dasyuridae						
Brush-tailed phascogale	*Phascogale tapoatafa*	0	3	3	1.73	
Common planigale	*Planigale maculata*	1	0	1	-1.00	-
Northern quoll	*Dasyurus hallucatus*	2	18	18	4.00	*<0.001*
Red-cheeked dunnart	*Sminthopsis virginiae*	0	1	1	1.00	-
Petauridae						
Sugar glider	*Petaurus breviceps*	5	0	5	-2.24	-
Medium sized mammals						
Carnivora						
Felidae						
Cat	*Felis catus*	2	5	7	1.13	-
Canidae						
Dingo	*Canis lupus dingo*	1	22	23	4.38	*<0.001*
Marsupialia						
Phalangeridae						
Common brushtail possum	*Trichosurus vulpecula*	5	9	10	1.63	*0.0547*
Macropodidae						
Godman's rock wallaby	*Petrogale godmani*	0	1	1	1.00	-
Rufous bettong	*Aepyprymnus rufescens*	0	1	1	1.00	-
Peramelidae						
Northern brown bandicoot	*Isoodon macrourus*	0	11	11	3.32	*0.0016*
Suidae						
Suinae						
Pig	*Sus scrofa*	0	22	22	4.69	*<0.001*
Monotremata						
Tachyglossidae						
Short-beaked echidna	*Tachyglossus aculeatus*	0	8	8	2.83	*0.0020*

Nine of the 20 species detected using camera traps were detected on at least one of the 63 experimental plots on the first night of deployment (Figure 6). These were the cane field rat, northern brown bandicoot, fawn-footed melomys, common rock-rat, dingo, northern quoll, short-beaked echidna, common brushtail possum and pig. Of these, the northern quoll, cat, dingo and pig took up to 30 nights or longer to be detected on other plots. The minimum and maximum number of trap nights to detect all species during the experimental plot surveys was 30 and 37 trap nights respectively (Figure 7). The species accumulation curve for the minimum time to detect all species appears to rapidly increase until trap night three (60 percent of captures) and then changes (inflection point) to a much more gradual rate of increase. The species accumulation curve for the maximum time to detect all species appeared to gradually increase for the entire 35 camera trap nights with no obvious plateauing occurring.

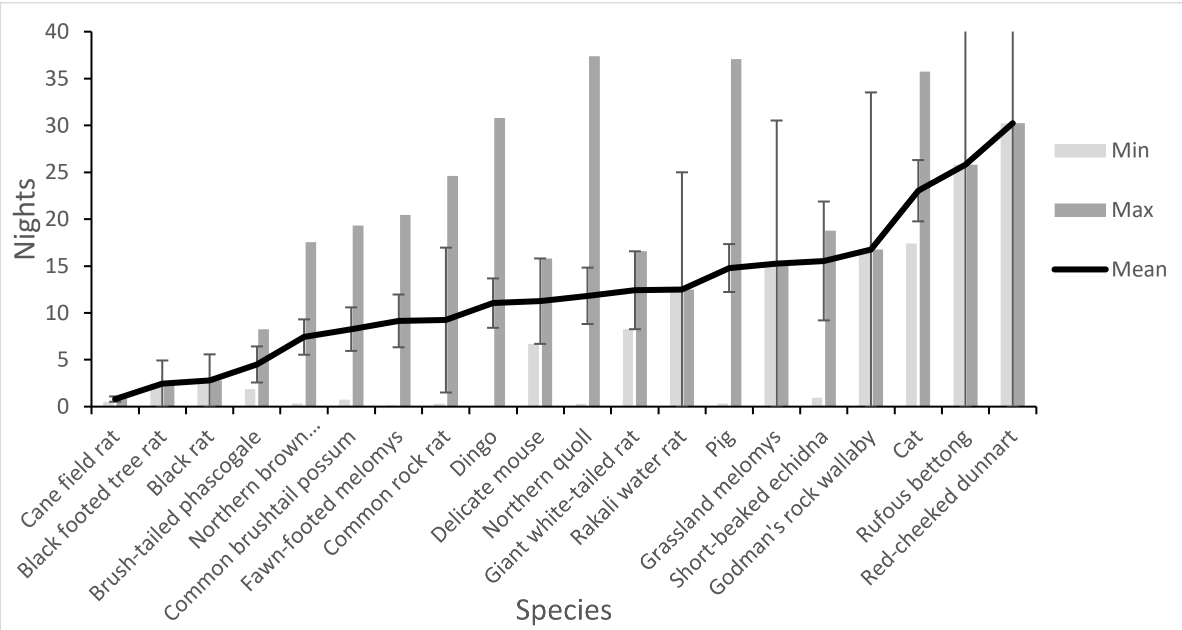

Figure 6. The minimum and maximum time (nights) to the first detection of small and medium sized native mammal species and large introduced vertebrate pests using camera trapping. Error bars indicate 95% confidence interval.

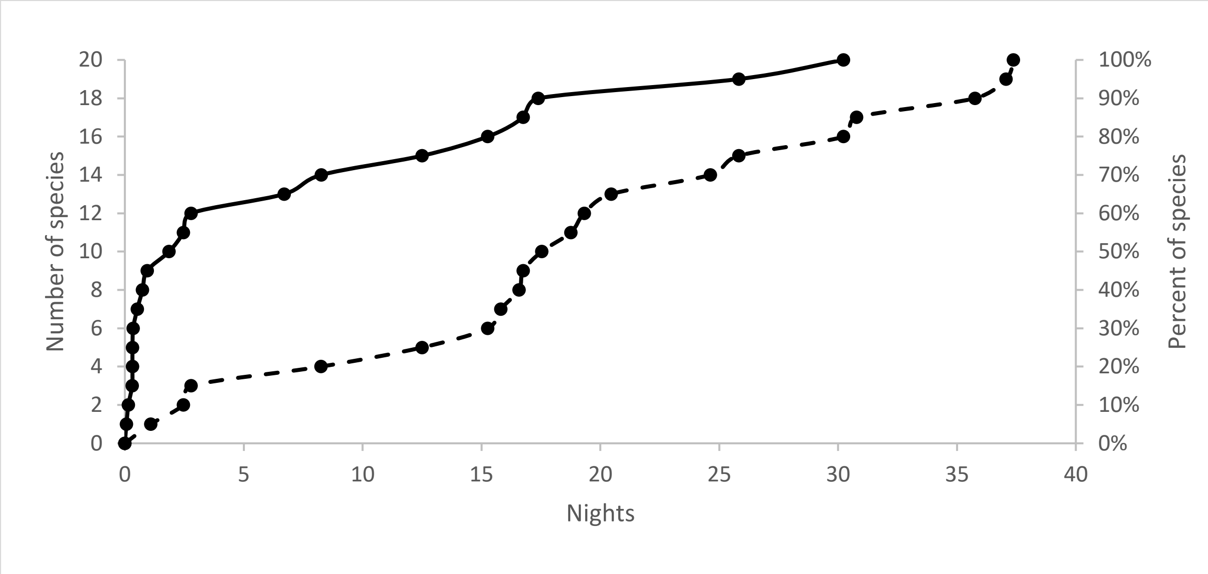

Figure 7. The minimum (solid line) and maximum time (dashed line) (nights) to detect all 20 mammal species using camera traps across 63 survey sites in North Queensland.

3.4 DISCUSSION

3.4.1 Plot surveys

The relative abundance and species richness of small and medium sized mammals identified in the Northern Gulf region is considerably lower than those reported in other studies across the top end of other parts of Northern Australia. For example, overall capture rates at Mornington Station were 6.08 per 100 trap nights (Legge *et al.* 2011) and for species which also occur in the Northern Gulf region and in Kakadu (e.g. Common planigale, Northern brown bandicoot, Pale field-rat), our capture rates in the Northern Gulf region were lower, except for the common brushtail possum (Woinarski *et al*. 2010). Mean abundance recorded by Woinarski *et al.* (2010) between 2007 -2009 (when declines were recorded) for the Common planigale, Northern brown bandicoot and Pale field-rat were 0.04, in comparison, abundances for these species were 0.03, 0.01 and 0,03 respectively in the Northern Gulf region. Elliott and pitfall trap success during the plot surveys was also considerably lower than recorded on two properties in the Northern Kimberley, where trapping was undertaken between 2006-2014 (Olds et al. 2016; Olds et al. 2017), which occurred during a similar time frame to this study 2003-2016. For example, Olds *et al.* (2016) recorded 2.7 captures per 100 trap nights for Elliott traps and 1.6 captures per 100 trap nights for pitfall traps whereas our study recorded 0.35 and 0.57 captures per 100 trap nights for Elliott and pitfall traps respectively. It is uncertain if mammals have declined (due to a lack of historic sampling in this region) or if these animals are naturally absent or at low abundance and richness. The region has experienced floods, extreme wildfires and drought in recent years, and it is plausible these impacts have acted synergistically with booming feral animal numbers such as feral cats and long-term cattle grazing to cause a decline in small and medium sized mammals.

Surveys in Blackbraes National Park and its neighbouring pastoral property in the Einasleigh Uplands bioregion yielded the highest abundance than elsewhere surveyed in the region. When these sites are excluded from the Einasleigh bioregion dataset, both abundance and species richness is similar to that observed in the Gulf Plains bioregion. The area is a rare representation of upland savanna where some mammal species declining elsewhere in northern Australia persist at high abundance (Vanderduys et al. 2012). The area contains heart-leaf poison-bush, *Gastrolobium grandiflorum*, a plant toxic to cattle, which has prevented intensive cattle grazing. These sites may provide important refugia, and future surveys are required to determine if other similar populations may persist in the Northern Gulf management region. Ecologically similar unsurveyed upland areas that also contain heart-leaf poison-bush may occur within the Gulf Plains bioregion (e.g. along the Gregory Range and west of Blackbraes National Park). Future surveys within these locations may identify other potential refugia for mammals in the Northern Gulf region.

3.4.2 Experimental surveys incorporating camera traps

Incorporating camera trapping into the plot survey methodologies in this study markedly increased the detection of several small and medium sized terrestrial mammal species. Five small sized and eight medium sized mammal species were exclusively detected with the use of camera traps. However, five small mammal species were only detected using the traditional plot survey methodologies. Three of these species were rodents (Eastern chestnut mouse, House mouse, Lakeland Downs mouse) and it is plausible that these species were detected using camera traps, however, were unidentifiable.

Camera traps were especially effective at detecting medium sized mammals, as all species were detected using camera traps. For surveys targeting medium sized mammals, camera traps could be used as a sole means of detection. One of the limitations of using only camera traps is determining accurate density of species. Mark and recapture techniques are not able to be used while camera trapping unless the species has physical features that can be used to identify individuals (e.g. using the spot pattern for identifying quolls). A less accurate index of relative abundance can be calculated using time between camera activations for species where individuals cannot be distinguished.

Incorporating camera trapping into the plot survey methodologies markedly increased the detection of several small and medium sized terrestrial mammal species. Northern quolls were detected using camera traps on 18 of the 63 plots, and a total of 50 camera events, yielding the highest camera trap success and distribution across plots of all native species in this study. The northern quoll is listed as endangered under the *Environment Protection and Biodiversity Conservation Act 1999* (*EPBC* Act) as the species has declined across most of north eastern Australia due to the introduction of the cane toad (Burnett 1997; Woinarski et al. 2008; Woinarski et al. 2014; Woinarski et al. 2015). Prior research indicates the northern quoll may have persisted in areas of refuge from toads such as upland rocky terrain (Woinarski et al. 2008; Hill & Ward 2010). Both the northern quoll and cane toad were found to be widely distributed across the experimental plots in the Cape York Management Region, including lowland woodlands and seasonally inundated grassy plains where quolls have presumably declined. Further, some evidence of quolls avoiding cane toads during this study was observed on camera images (Appendix 5) suggesting the species has developed prey avoidance strategies and now occur sympatrically with cane toads. Camera traps set with the lens facing downward are particularly useful for surveying northern quoll populations by using the unique pattern of spots on the dorsal surface to identify individuals (Hohnen et al. 2013; Diete et al. 2015; Webb et al. 2015; Austin et al. 2017). However, our camera traps were set vertically for sampling a wider range of species and the images were not useful for identifying individual quolls.

The northern brown bandicoot and short-beaked echidna were detected on significantly more plots by camera trapping than the standard plot survey methodologies. Both species were detected exclusively using camera traps on a relatively large number of plots, 11 and 8 respectively. The northern brown bandicoot, a species considered to be declining across parts of northern Australia

(Braithwaite & Muller 1997; Woinarski et al. 2001; Pardon et al. 2003), failed to be detected on the experimental plots using traditional survey methodologies, despite the species being readily captured using cage traps and spotlighting in other studies (Hall 1983; Woinarski et al. 2004; O'Hara 2014). Small numbers of individuals were detected using cage traps (5), Elliott traps (1) and spotlighting (1) in the plot surveys between 2003 and 2016. A similar study by De Bondi et al. (2010) comparing camera traps and live trapping, detected the closely related southern brown bandicoot, *Isoodon obesulus*, exclusively using camera traps. Trap aversion or cryptic behaviour may have contributed to the lack of captures using standardised plot methodologies as detection using cameras in some plots occurred on the first night of camera trapping. Camera traps therefore, may provide a useful tool to enhancing detection success for this species across the Northern Gulf Management Region.

The short-beaked echidna being exclusively detected by camera traps on the experimental plots is not surprising as the species is not often detected using standard plot survey methodologies. In north Queensland the species is only rarely captured in pitfall traps or observed during night active searches (e.g., three individuals were observed during spotlighting on 430 plots in the Northern Gulf Management region during this study). Echidnas in Northern Australia are predominantly nocturnal as the species lacks the physiological mechanisms for thermoregulation and therefore rely on behavioural mechanisms to regulate body temperatures in high ambient temperatures (Brice et al. 2002a). In this study short-beaked echidnas were detected during night hours between 2205 h and 0510 (dawn). A similar active period for echidnas has been reported in south-west Queensland in summer where daytime temperatures range between 33°C and 40°C (Brice et al. 2002b).

Eight native mammal species were detected exclusively by camera traps during the experimental plot surveys and five of these were detected on only one plot. It is plausible that these species occur at low densities, have a small distribution, or are elusive. The Rakali water rat, rufous bettong and red-cheeked dunnart were detected by cameras only on 12th, 25th and 30th trap nights respectively, suggesting these species may be relatively trap averse. We therefore recommend increasing the duration of camera trap deployment for detecting these rare or elusive species (Gálvez et al. 2016).

The standard plot survey methodologies detected the grassland melomys on significantly more plots than camera traps during the experimental plot surveys. This is surprising given the results were not significant for the closely related fawn-footed melomys. The only plausible reason for the species being undetected by camera traps is a reluctance to venture out from vegetative cover into clearing around the lure. When setting camera traps, vegetation is cleared between the camera trap and lure to ensure vegetation does not interfere with data collection and cause thousands of misfired images. The species was captured predominantly with Elliott traps, which are generally set under vegetation cover.

Although we were not able to identify many rodents to genus or species level in camera trap photographs in this study, it is unlikely many of these were unidentified grassland melomys. Unidentifiable rodents were only recorded on one of the seven sites where grassland melomys were detected using standardised plot survey methodologies, and this was on a site where the grassland melomys was detected only with the use of spotlighting. It is also unlikely that the grassland melomys was misidentified as the fawn-footed melomys as the two species occupy different habitat types (Kerle 2004; Redhead 2004).

Camera trapping in this study was particularly effective in detecting introduced species as bycatch (four of the five introduced species were detected by camera traps). Camera traps are increasingly being used to monitor the abundance of pest animal populations in response to control programs (Cowled et al. 2006; Robley et al. 2008; Robley 2010; Bengsen et al. 2011; Bengsen 2014). Camera trapping is especially effective in monitoring cat and pig populations as individuals can be identified using coat patches, colour, sex and relative size (Cowled et al. 2006; Bengsen 2014). Incorporating camera traps into fauna survey methodologies provides valuable information on pest species distribution and abundance and may be useful for determining the influence these pest species, particularly cats, have on small and medium sized native mammal populations. For effective survey of cat populations, we recommend increasing the number of cameras used concurrently at each survey site and extending the duration of camera deployment (see Stokeld et al. 2015).

Nine of the 20 species detected by camera traps in this study were detected using camera traps on at least one of the 36 plots on the first night of deployment. However, many of these species had highly variable rates until first detection across the plots. For example, the northern quoll was first detected on one plot on the 37th night of deployment. Increasing camera trap deployment rates may reduce the chance of surveys making a type II error for some species.

3.4.3 Management implications

Our data have clearly shown that the relative abundance and species richness of small and medium sized mammals is lower in the Northern Gulf management region than other parts of northern Australia. It is not known if these mammals have declined in the Northern Gulf management region given there is no long-term historic data in the region. If populations in northern Queensland are undergoing similar declines reported elsewhere in the top end, urgent actions may be required to address these declines by assessing their conservation status more thoroughly across the region. An ongoing small mammal monitoring program, incorporating camera traps into the recommended state guidelines, should be undertaken in Blackbraes National Park and the surrounding properties as these appear to provide important refugia for small and medium sized mammals.

Incorporating camera trapping into the plot survey methodologies may improve our knowledge of the distribution of these small and medium sized mammals, particularly for rare and elusive species, in the Northern Gulf management region. Camera trapping may also be used to assess the conservation status of some species considered to be declining across northern Australia (eg. northern quoll, northern brown bandicoot and black-footed tree rat). In conclusion, adding camera trapping with increased deployment times (up to 37 days) to the traditional fauna survey methodologies will assist with improving the effectiveness of fauna surveys in detecting small and medium sized mammals.

3.5 CONTRIBUTION TO AUTHORSHIP

The following outlines my contribution to the authorship of this chapter in the thesis:

I made substantial contribution to the conception and design of the project. I collated all historical data (provided by Northern Gulf Resource Management Group) and was a key participant on all subsequent field surveys where data was collected. I received some assistance from Dr. Carly Starr with project design and organising the field surveys.

I drafted the entire chapter and developed all the maps and figures. I undertook the statistical analysis with assistance from Dr. Allan Lisle. Interpretation of the research data was undertaken with some assistance from Dr. Luke Leung. I wrote the entire discussion which was then edited by my supervisor Dr. Luke Leung.

CHAPTER 4: CAMERA TRAPPING IN NORTHERN QUEENSLAND: DO LONGER-LIFE ATTRACTANTS IMPROVE DETECTION RATES OF SMALL AND MEDIUM SIZED MAMMALS IN TROPICAL SAVANNA?

4.1 INTRODUCTION

The use of camera traps in fauna monitoring and research in Australia has become increasingly popular in recent years (Burton et al. 2015; Meek et al. 2015b). Camera traps offer many advantages over traditional fauna survey techniques that rely on live-capture of animals. Several studies have proven the effectiveness of camera trapping in detecting mammals in Australia, particularly when camera traps are deployed for longer periods (Claridge et al. 2010; De Bondi et al. 2010; Paull et al. 2011; Paull et al. 2012; Swan et al. 2014). Camera traps have been found to be particularly useful for detecting animals at low abundance or rare or cryptic species that may not be easily detected using live trapping (Claridge et al. 2015; McDonald et al. 2015). Northern Queensland is relatively remote, and some small mammals are at low abundance in tropical savanna (Starr & Waller 2013a, 2013c; Starr et al. 2015a; Starr et al. 2015b; Starr & Waller 2016; Starr et al. 2017). Integrating camera trapping into systematic fauna surveys by the authors has been successful in improving detection of several mammal species in northern Queensland, yet the detection rates remain low. Further, camera trapping detected low numbers of several native mammal species (black-footed tree rat, brush-tailed phascogale, *Phascogale tapoatafa*, Godman's rock wallaby, *Petrogale godmani*, rakali water rat, *Hydromys chrysogaster,* red-cheeked dunnart, *Sminthopsis virginiae*, rufous bettong, *Aepyprymnus rufescens*, short-beaked echidna) that were undetected by other fauna survey methodologies.

It is generally considered that the longer camera traps are deployed the better the detection rate (Kays et al. 2009; Meek et al. 2012). Suggested camera trap deployment times for detecting small and medium sized mammals in northern Australia range from 14 nights to five weeks (Paull et al. 2011; Meek et al. 2012; Gillespie et al. 2015).

Camera traps can be used passively or baited with a lure to attract animals into the field of view (Rovero et al. 2013). Baited camera traps have been found by several studies in Australia to increase trap success by luring a passing animal into the detection zone (Paull et al. 2011; Diete et al. 2015; Austin et al. 2017). Increasing the time an animal spends while investigating bait stations also provide greater opportunities for identification of species or individuals where unique physical features may be evident (Hohnen et al. 2013; Diete et al. 2015).

A handful of studies have critically evaluated bait attractiveness for camera trapping wildlife in Australia (Paull et al. 2011; Claridge et al. 2015; Diete et al. 2015; Austin et al. 2017). However, no study has previously been undertaken to improve or test bait longevity. When collecting camera traps in tropical savanna, after a 30 day period, bait is often missing, partially consumed by invertebrates, decomposed or mouldy, despite being contained in a plumbing cowl for protection. Wax block baits have been used successfully in rodent control programs due to their environmental longevity and practicability in the field (Connor & Eason 2000). Non-toxic wax block bait may provide a longer life attractant for the duration required for camera trapping in tropical savanna (≥ 20 days).

The primary aim of this study is to evaluate the effectiveness of two wax block baits for camera trapping small and medium sized mammals in tropical savanna. In this study we also aim to quantify the longevity of wax block baits for deployment in tropical savanna.

4.2 METHODS

4.2.1 Study site

The trial was undertaken at the Mareeba Tropical Savanna and Wetland Reserve Nature Refuge (MTSWRNR) (16°55’27.8”S, 145°21’37.1”E), Biboohra, Queensland (Figure 8). Data were collected along the main access track in the north-east section of the refuge within eucalypt woodland, primarily in the broad vegetation group “Low woodlands and low open woodlands dominated by *Melaleuca viridiflora* (coarse-leaved paperbark) on depositional plains”.

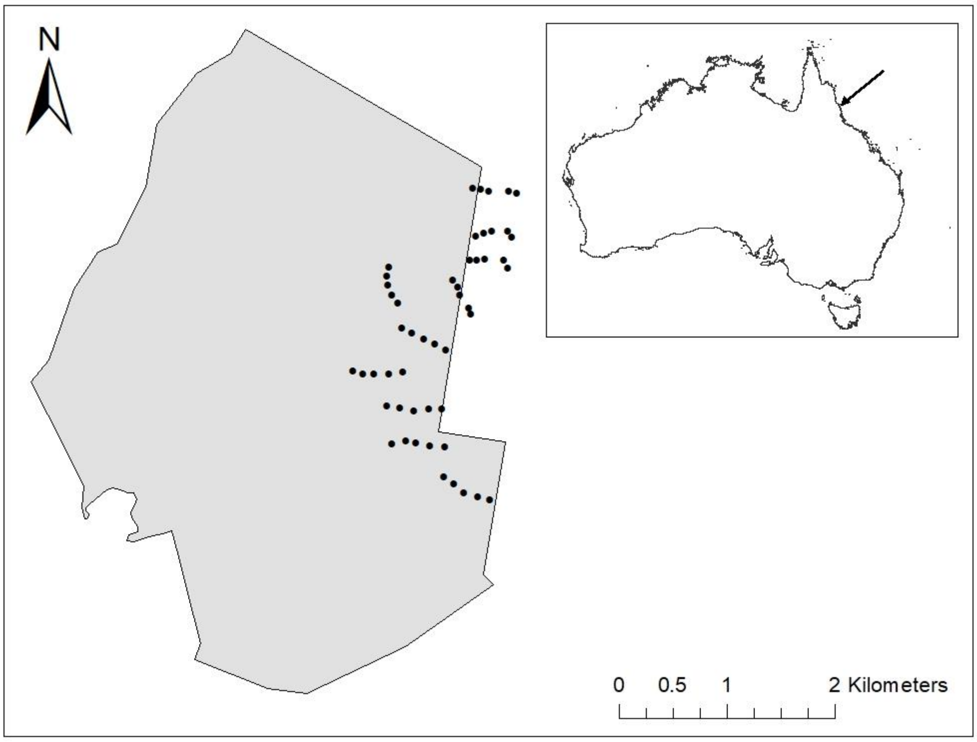

Figure 8. Location of the camera traps (black dots) at the Mareeba Tropical Savanna and Wetland Reserve Nature Refuge, Biboohra, Queensland. The location of the study site is indicated in the inserted map of Australia.

4.2.2 Baits

Two newly formulated wax block baits were tested for their attractiveness against standard peanut butter bait (PB) commonly used in ecological studies (Menkhorst & Knight 2011). The two trial baits were sesame oil (10% by weight) infused into bees wax (S) and peanut butter (10%), vanilla essence (5%) and oats (5%) infused into bees wax (PW). These formulations were based on the effectiveness of the constituents in recent camera trapping bait experiments, particularly at attracting northern quolls, bandicoots and rodents (Paull et al. 2011; Claridge et al. 2015; Diete et al. 2015; Austin et al. 2017) all of which are known to occur at the study site. Bees wax was used as a moisture barrier medium to prolong the longevity of baits. The bait formulations were melted into ice cube trays and left to harden for 24 hours. A wax block bait weighing 50 g was then placed into a cylindrical PVC plumbing cowl (110 x 65 mm) for protection against disturbance and consumption during the trials.

4.2.3 Trial design

Ten transects (>200 m apart) each consisting of five RECONYX Hyperfire HC550 camera traps (RECONYX Inc., Holmen, WI) placed 50 metres apart, were set across the north eastern section of the refuge. The study was undertaken between January and March 2016 in three trial periods. During the first trial period, each camera trap was set horizontally directed at a plumbing cowl holding one of the three bait types chosen randomly. The plumbing cowl was placed 1.5 to 2 m from the camera trap. Camera traps were set at high sensitivity to take three images per trigger event, with no quiet period. Camera traps were positioned ~35–40 cm above the ground, with the bait located within the centre of the field of view between. Each bait type was rotated every 30 days using a latin square cross over design through each camera trap during the 90 days of the trial period. Cameras were deployed for approximately 21 days during each trial period, with at least one week wash-out period between trials. The time of each trial period and the bait placed at each station is shown in Table 6. Not all camera traps were fully functional during the study. During trial 1, camera 35 and 49 malfunctioned and during trial 3, camera 16 malfunctioned and camera 47 was missing as it was removed during the trial period.

4.2.4 Event definitions

The attractiveness of bait was quantified by the number of photographic events (visits) for a species or taxon. An event is defined as one or a sequence of consecutive photographs of an individual of no more than a species-specific interval apart. The probability of an event caused by the recapture of an individual should be equal or no greater than that caused by a newly captured individual. This species-specific interval was determined with photographic data of known individuals identified by natural marking to be 15 minutes for the northern quoll and northern brown bandicoot and, 30 minutes for the agile wallaby by Diete *et al.* (Diete et al. 2015). For all the other study species this interval was determined by examining the data of the present study.

Table 6. The time period and baits tested at each bait station during the three trial periods at Mareeba Wetlands

STATION #	TRIAL PERIOD 1 11/1/16 – 1/2/16	TRIAL PERIOD 2 11/2/16 – 3/3/16	TRIAL PERIOD 3 9/3/16 – 30/3/16
1	Peanut butter	Sesame oil wax	Peanut butter wax
2	Peanut butter wax	Peanut butter	Sesame oil wax
3	Sesame oil wax	Peanut butter wax	Peanut butter
4	Peanut butter	Sesame oil wax	Peanut butter wax
5	Peanut butter wax	Peanut butter	Sesame oil wax
6	Sesame oil wax	Peanut butter wax	Peanut butter
7	Peanut butter wax	Peanut butter	Sesame oil wax
8	Peanut butter wax	Peanut butter	Sesame oil wax
9	Sesame oil wax	Peanut butter wax	Peanut butter
10	Peanut butter wax	Peanut butter	Sesame oil wax
11	Peanut butter	Sesame oil wax	Peanut butter wax
12	Peanut butter wax	Peanut butter	Sesame oil wax
13	Peanut butter	Sesame oil wax	Peanut butter wax
14	Peanut butter wax	Peanut butter	Sesame oil wax
15	Sesame oil wax	Peanut butter wax	Peanut butter
16	Peanut butter wax	Peanut butter	malfunctioned
17	Sesame oil wax	Peanut butter wax	Peanut butter
18	Peanut butter	Sesame oil wax	Peanut butter wax
19	Peanut butter wax	Peanut butter	Sesame oil wax
20	Sesame oil wax	Peanut butter wax	Peanut butter
21	Peanut butter wax	Peanut butter	Sesame oil wax
22	Sesame oil wax	Peanut butter wax	Peanut butter
23	Peanut butter wax	Peanut butter	Sesame oil wax
24	Sesame oil wax	Peanut butter wax	Peanut butter
25	Peanut butter	Sesame oil wax	Peanut butter wax
26	Sesame oil wax	Peanut butter wax	Peanut butter
27	Peanut butter	Sesame oil wax	Peanut butter wax
28	Peanut butter wax	Peanut butter	Sesame oil wax
29	Sesame oil wax	Peanut butter wax	Peanut butter
30	Peanut butter wax	Peanut butter	Sesame oil wax
31	Peanut butter	Sesame oil wax	Peanut butter wax
32	Peanut butter wax	Peanut butter	Sesame oil wax
33	Peanut butter wax	Peanut butter	Sesame oil wax
34	Sesame oil wax	Peanut butter wax	Peanut butter
35	malfunctioned	Sesame oil wax	Peanut butter wax
36	Peanut butter wax	Peanut butter	Sesame oil wax
37	Peanut butter	Sesame oil wax	Peanut butter wax
38	Peanut butter	Sesame oil wax	Peanut butter wax
39	Peanut butter wax	Peanut butter	Sesame oil wax
40	Sesame oil wax	Peanut butter wax	Peanut butter
41	Peanut butter wax	Peanut butter	Sesame oil wax
42	Sesame oil wax	Peanut butter wax	Peanut butter
43	Sesame oil wax	Peanut butter wax	Peanut butter
44	Sesame oil wax	Peanut butter wax	Peanut butter
45	Peanut butter	Sesame oil wax	Peanut butter wax
46	Peanut butter	Sesame oil wax	Peanut butter wax
47	Removed	Removed	Removed
48	Sesame oil wax	Peanut butter wax	Peanut butter
49	malfunctioned	Sesame oil wax	Peanut butter wax
50	Peanut butter	Sesame oil wax	Peanut butter wax

The time interval set for the common brushtail possum and rufous bettong were both 30 minutes. Of 27 photographic events separated by an interval of less than 30 minutes, three were of different individuals of the common brushtail possum, 21 were of the same individual, and two could not be confidently distinguished between these two categories. Of nine events between 30 and 60 minutes apart, four were of different individuals of the common brushtail possum, three were of the same individual and two could not be confidently distinguished between these two categories. Individuals were recognised by distinctive morphological features such as ear notches, colouration, scars or body size.

Of 17 events separated by an interval of less than 60 minutes there were seven instances where successive images of rufous bettongs less than 60 min apart could be distinguished as being of different individuals. Individuals could be recognised for this species by distinctive characterisations such as ear notches, colouration, injuries or body size. Of the seven instances, three occurred less than 15 minutes since the previous detection of the species and the remainder were between 26 and 60 minutes apart. Based on this anecdotal evidence, we selected 30 minutes as the event delineator for this species. Due to the low number of occurrences of rodents, 60 minutes was set as the event definition for this species group. In no instances did rodents visit the bait stations on the same date. Definitions were set as 30 minutes for reptiles and 60 minutes for birds and amphibians.

4.2.5 Bait longevity

Bait longevity was scored when removing bait from the canister at the end of each trial period: no mould present; slightly mouldy; or very mouldy. If bait was missing it was classified as missing.

4.2.6 Statistical analysis

An analysis of variance (ANOVA) was used to determine if bait type had a significant effect on mean number of events and on species richness. The data were transformed by log (x+1) for normality. The fitted means were back-transformed for presentation. The factors in the analysis were camera points (n = 49), period (3) and bait type (3). An ANOVA was also used to determine if bait type significantly affected the species richness of mammal groups identified at Mareeba wetlands. Data for analysis were in the following groups: native mammals (all native mammal species detected); Introduced species (dog, cat and pig); small native mammal species (native mammals 0 to 5 kilograms in weight); and macropods (Table 7.). The cow and horse were removed from any statistical analysis, as these species are actively grazed on the property. Tukey tests were used for post hoc pairwise comparisons.

The association between bait type and presence of mould was analysed with a Chi-square test using R Studio version 1.0.143.

Table 7. List of species in each group for analysis.

Group	Species
Native mammals	Agile Wallaby *Notamacropus agilis,* Common brushtail possum *Trichosurus vulpecula,* Common wallaroo *Macropus robustus,* Eastern grey kangaroo *Macropus giganteus,* Unidentifiable Macropod sp. *Macropus spp.,* Northern brown bandicoot *Isoodon macrourus,* Northern quoll *Dasyurus hallucatus,* Rufous Bettong *Aepyprymnus rufescens,* Short-beaked echidna *Tachyglossus aculeatus.*
Introduced species	Cat *Felis catus*, Wild dog *Canis lupus dingo*, Pig *Sus scrofa*.
Small native mammals	Common brushtail possum *Trichosurus vulpecula,* Northern brown bandicoot *Isoodon macrourus,* Northern quoll *Dasyurus hallucatus,* Rufous Bettong *Aepyprymnus rufescens,* Short-beaked echidna *Tachyglossus aculeatus.*
Macropods	Agile Wallaby *Notamacropus agilis,* Common wallaroo *Macropus robustus,* Eastern grey kangaroo *Macropus giganteus,* Unidentifiable Macropod sp. *Macropus spp.,* Rufous Bettong *Aepyprymnus rufescens.*

4.3 RESULTS

4.3.1 Bait attractiveness

Not all camera traps were fully functional during the study. During trial 1, camera 35 and 49 malfunctioned and during trial 3, camera 16 malfunctioned and camera 47 was missing as it was removed during the trial period. Sampling efforts totalled 2,776 camera trap nights (CTN) with 16,531 images and 1269 events collected. Of these events, 1269 events were of animals that could be identified to species, genus or order level. A total of 39 species were identified in these events including 13 mammal species (Table 8).

Bait type significantly affected mean camera events for the Agile wallaby ($F_{2,48}$ = 3.37, P = 0.039) (Table 9 and Figure 9) with the sesame oil wax bait stations being visited 1.5 times more than peanut butter wax bait (P = 0.030). There was no significant difference between sesame oil wax bait and peanut butter bait (P = 0.568). Bait type did not significantly affect the mean number of events for any of the other species of mammals detected during this study (Table 9).

Table 8. Summary of the species captured on camera traps during the three trial periods from 50 camera stations set in 10 transects.

Common name	Scientific name	Events
Mammals		
Agile Wallaby	*Notamacropus agilis*	270
Cat	*Felis catus*	3
Common Brushtail Possum	*Trichosurus vulpecula*	250
Common wallaroo	*Macropus robustus*	5
Cow	*Bos Taurus*	72
Wild dog	*Canis lupus dingo*	4
Eastern grey kangaroo	*Macropus giganteus*	22
Horse	*Equus caballus*	17
Macropod sp.	*Macropus spp.*	29
Northern brown bandicoot	*Isoodon macrourus*	42
Northern quoll	*Dasyurus hallucatus*	11
Pig	*Sus scrofa*	7
Rodent sp.	*Rodentia spp.*	4
Rufous Bettong	*Aepyprymnus rufescens*	286
Short-beaked echidna	*Tachyglossus aculeatus*	9
unidentifiable mammal sp.	Mammal spp.	33
Birds		
Apostlebird	*Struthidea cinerea*	13
Bar-shouldered dove	*Geopelia humeralis*	4
Black-throated finch	*Poephila cincta*	1
Blue-faced honeyeater	*Entomyzon cyanotis*	1
Brown goshawk	*Accipiter fasciatus*	1
Bush stone curlew	*Burhinus grallarius*	5
Common bronzewing	*Phaps chalcoptera*	3
Dollarbird	*Eurystomus orientalis*	1
Double-barred finch	*Taeniopygia bichenovii*	1
Emu	*Dromaius novaehollandiae*	32
Great bowerbird	*Chlamydera nuchalis*	3
Grey shrike-thrush	*Colluricincla harmonica*	2
Grey-crowned babbler	*Pomatostomus temporalis*	4
Laughing kookaburra	*Dacelo novaeguineae*	3
Lovely fairy wren	*Malurus amabilis*	1
Australian magpie	*Cracticus tibicen*	2
Magpie-lark	*Grallina cyanoleuca*	3
Painted button quail	*Turnix varius*	14
Peaceful dove	*Geopelia placida*	21
Pheasant coucal	*Centropus phasianinus*	36
Pied butcherbird	*Cracticus nigrogularis*	8
Rufous songlark	*Megalurus mathewsi*	1
Tawny frogmouth	*Podargus strigoides*	1
Unidentifiable bird sp.	*Aves spp.*	3
Wedge-tailed eagle	*Aquila audax*	1
Amphibians		
Cane toad	*Rhinella marina*	5
Reptiles		
Frilled lizard	*Chlamydosaurus kingii*	35

Table 9. The mean number of events per bait type over 21 days, for species detected using camera trapping at Mareeba Tropical Savanna and Wetland Reserve Nature Refuge. PB = Peanut butter bait, PW = Peanut butter wax bait. S = Sesame oil wax bait.

		Bait type			
Common name	**Scientific name**	**PB**	**PW**	**S**	**P-value**
Agile Wallaby	*Notamacropus agilis*	1.826	1.588	2.143	*0.039*
Cat	*Felis catus*	0.000	0.019	0.041	*0.566*
Common Brushtail Possum	*Trichosurus vulpecula*	2.195	1.706	1.245	*0.081*
Common wallaroo	*Macropus robustus*	0.022	0.019	0.061	*0.851*
Wild dog	*Canis lupus dingo*	0.043	0.019	0.020	*0.571*
Eastern grey kangaroo	*Macropus giganteus*	0.163	0.117	0.163	*0.576*
Unidentifiable Macropod sp.	*Macropus spp.*	0.130	0.274	0.183	*0.647*
Northern brown bandicoot	*Isoodon macrourus*	0.369	0.235	0.265	*0.362*
Northern quoll	*Dasyurus hallucatus*	0.065	0.098	0.061	*0.633*
Pig	*Sus scrofa*	0.021	0.058	0.061	*0.569*
Unidentifiable Rodent sp.	*Rodentia spp.*	0.000	0.039	0.040	*0.648*
Rufous Bettong	*Aepyprymnus rufescens*	2.500	2.039	1.367	*0.262*
Short-beaked echidna	*Tachyglossus aculeatus*	0.065	0.098	0.020	*0.154*

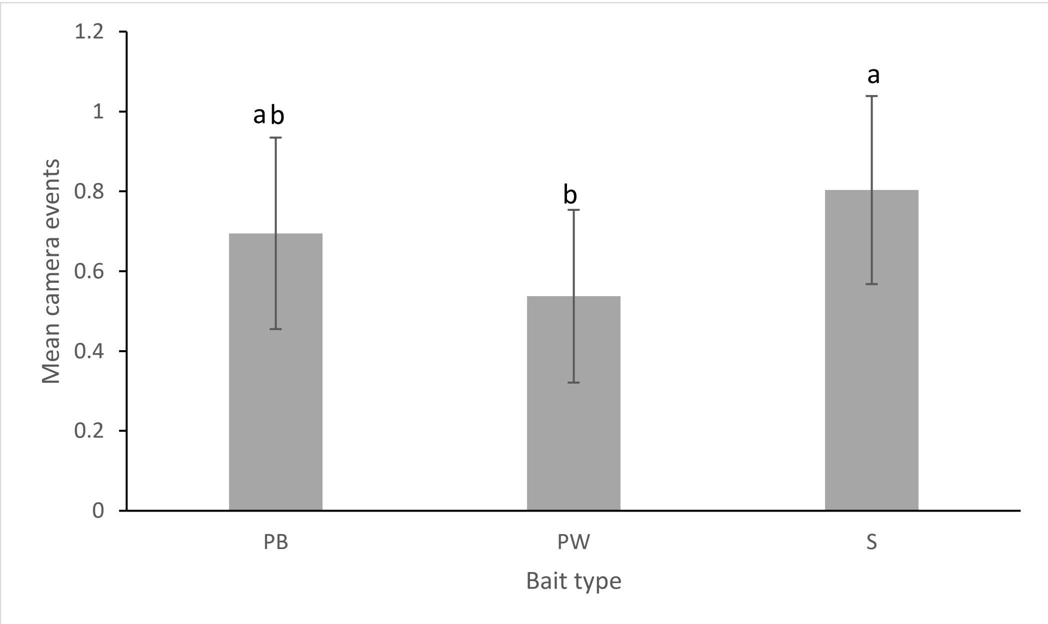

Figure 9. The mean number of camera events for the agile wallaby. Error bars represent 95% confidence intervals. PB = Peanut butter bait, PW = Peanut butter wax bait. S = Sesame oil wax bait. Columns with different letters are significantly different as determined by Tukey pairwise comparisons.

Bait type significantly affected mean species richness of small native mammal species ($F_{2,48}$ = 5.97, $P = 0.004$), but not native mammal species richness ($F_{2,48}$ = 2.07, P = 0.132). Peanut butter bait attracted 1.7 times more small mammal species than sesame oil wax bait ($P = 0.003$) (Figure 10). There was no significant difference between peanut butter bait and peanut butter wax bait ($P = 0.234$) or between the attractiveness of peanut butter wax bait and sesame oil wax bait ($P = 0.156$) for small mammal richness. Bait type did not significantly affect the mean number of events for macropods ($F_{2,48}$ = 1.01, P = 0.369).

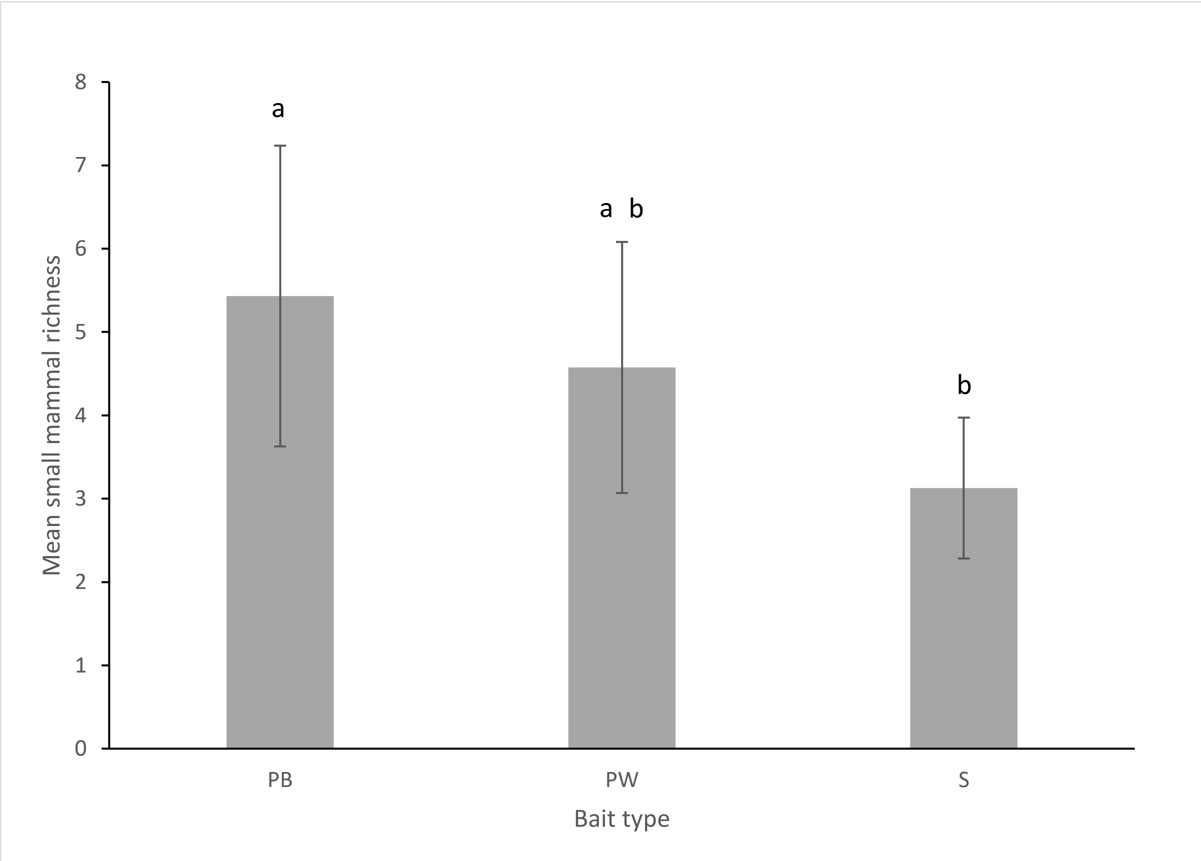

Figure 10. The mean species richness of small native mammal species detected using three bait types. Error bars represent 95% confidence intervals. PB = Peanut butter bait, PW = Peanut butter wax bait. S = Sesame oil wax bait. Columns with a different letter are significantly different as determined by Tukey pairwise comparisons.

The number of camera events for each bait type varied considerably across the course of the study (Figure 11). Sesame oil wax bait appeared to perform the most consistently over the entire 21 day period. However, sesame oil wax bait had lower number of camera events at the beginning of the trial period. Both peanut derived baits had higher number of camera events at the beginning of the trial period and events continued to drop over the durations of the survey period.

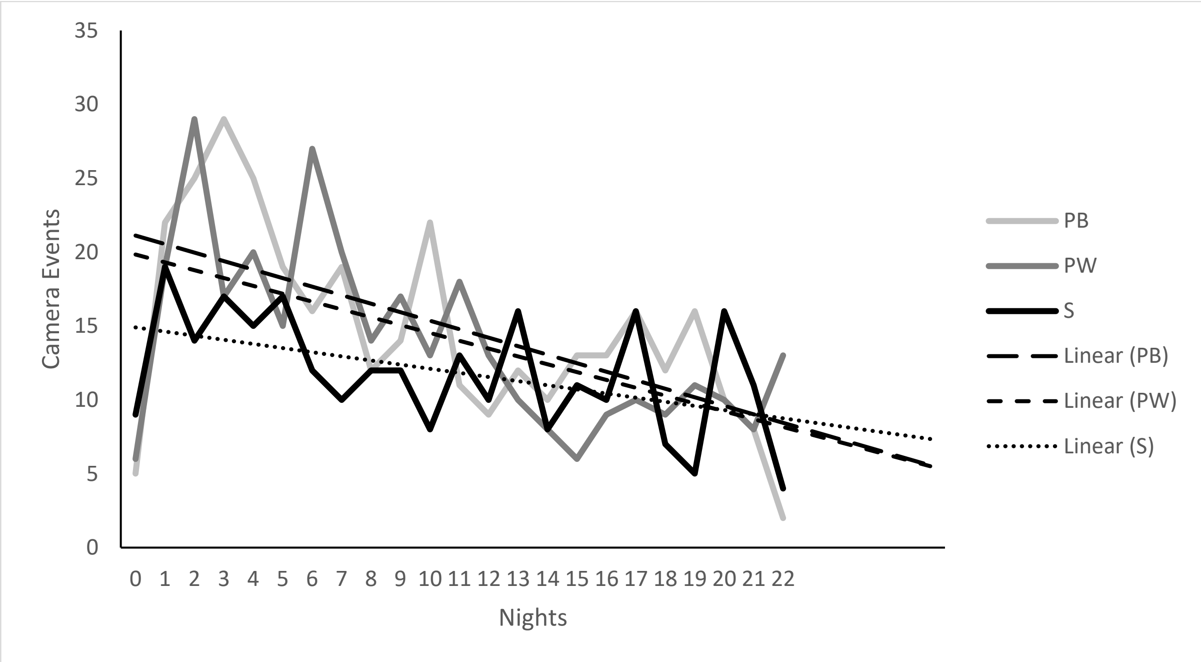

Figure 11. The number of camera trap events recorded for three bait types: Peanut butter bait (PB), Peanut butter wax bait (PW) and Sesame oil wax bait (S) over 22 consecutive days. Dotted lines are indicative of the modelled linear regression of the camera events over time for each bait type.

4.3.2 Bait longevity

Over the duration of the field trial, three baits were scored missing at end of the trial periods (3 PB, 1 PW). There was a significant association between bait type and mould (Chi-square = 132.47; degrees of freedom = 8; P = < 0.001). No sesame oil wax bait was mouldy when removed from the bait canisters. Forty-six of the forty-nine peanut butter baits were found to contain some mould at the end of the experiment (93.8%). Of these sixteen (34.8%) were slightly mouldy and thirty (65.2%) were considered very mouldy. Peanut butter wax baits only had twenty-two of the fifty baits containing some mould (44%) and all twenty-two baits with mould were considered slightly mouldy (Figure 12).

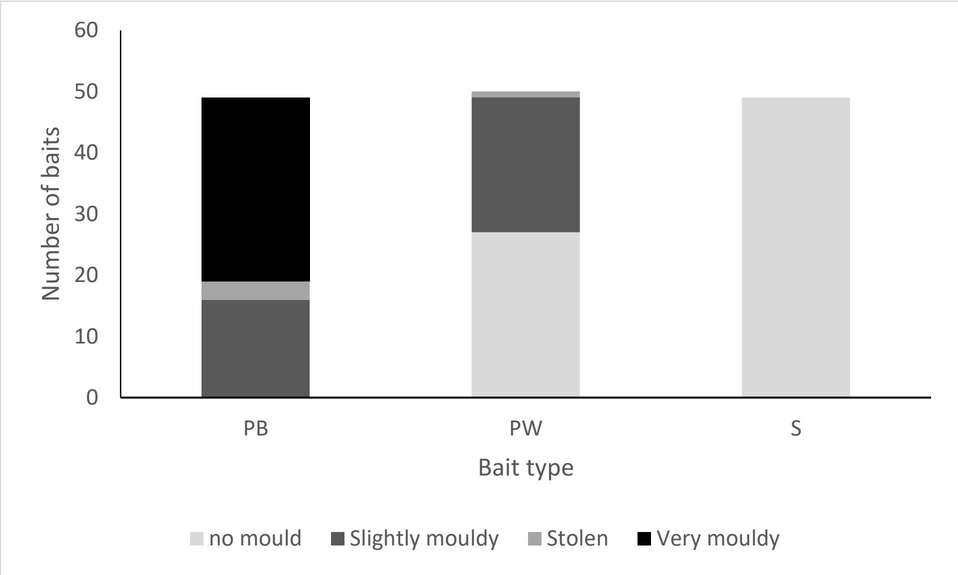

Figure 12. The decomposition condition of Peanut butter bait (PB), Peanut butter wax bait (PW) and Sesame oil wax (S) bait after deployment in the field for 21 days.

4.4 DISCUSSION

Both wax block baits and the standard peanut butter bait were effective at luring the same small and medium size mammal species at the Mareeba Tropical Savanna and Wetland Reserve Nature Refuge. Eight of the twenty-two native mammal species known to be found on the reserve (Appendix 6) were detected using these baits during this study. The most commonly detected mammal species were the rufous bettong, agile wallaby and common brushtail possum. All three species are relatively common across northern Queensland. However, the brushtail possum is considered declining in some locations (Winter 2007).

When considering bait attractiveness, standard peanut butter bait performed significantly better than the sesame oil wax bait in detecting small native mammal richness. The peanut butter bait is therefore more suited for detecting small native mammals in tropical savanna in northern Queensland. The standard peanut butter bait and peanut butter wax baits were comparable for detecting small native mammals. This is unsurprising given the constituents of the two bait types were relatively similar. Baits containing peanut butter are widely used to trap small mammals in Australia (Catling et al. 1997; Woinarski et al. 2010; Menkhorst & Knight

2011) and peanut butter and rolled oats is consistently found to be a superior bait for targeting a range of small mammal species in camera surveys (Paull et al. 2011; Claridge et al. 2015; Diete et al. 2015). Considering these results, using wax to prolong the life of the bait has not caused any reduction in the attractiveness of the bait for small native mammals. Furthermore, infusing peanut butter, rolled oats and vanilla essence into wax provides no benefits for detecting small native mammals over camera trapping periods less than 21 days in duration. However, a future trial with a larger sample size, over a longer duration than the present trial, should be conducted to critically determine the attractiveness of the peanut wax bait over longer periods.

Bait type significantly affected small native mammal species richness, but not native mammal species richness. The latter group included macropods, some of which appear to have higher mean events for sesame oil wax bait, particularly the agile wallaby. A slightly higher preference for sesame oil by some macropod species may explain why bait type did not affect native mammal species richness.

In this study, agile wallabies exhibited some preference for the sesame oil wax bait over peanut butter bait with peanut butter wax bait being the least attractive. Both sesame oil wax bait and standard peanut butter bait can therefore be considered suitable for detecting agile wallabies and may be used for camera trapping studies specific to this species. Our results for agile wallabies, however, contrast with a similar study undertaken by Diete et al. (2015), where no preference for bait type was detected. Diete et al. (2015) also found that agile wallabies were attracted to a control bait station containing no scent lure and spent a large amount of time interacting with the bait station or resting near bait stations. Most agile wallabies in this study spent considerable time investigating the bait station, and it is possible that they were investigating the bait station itself, rather than the scent lure inside. As an unbaited control was not used in this study and standard peanut butter bait acted as our control, it was not possible to determine if the agile wallabies were attracted to the bait or bait station.

We did not find preference for bait types in any of the other species in this study. Increasing the duration of camera deployment and/or sample size may improve the experimental power to find a significant difference between bait types for other species.

Surprisingly rodents were not detected by standard peanut butter bait. This might be possibly due to the small sample size caused by low abundance of rodents across the study site. Furthermore, a variety of rodent species have successfully been detected by camera traps baited with standard peanut butter bait in other locations in northern Queensland (Starr et al. 2015b; Starr et al. 2017). Peanut butter bait is considered superior to other baits for detecting rodents in camera surveys (Paull et al. 2012; Claridge et al. 2015).

Very few native carnivorous species were detected during this survey, the endangered northern quoll was detected by all three bait types during this study. Peanut butter and sesame-based baits have been found to lure high numbers of the species on Groote Eylandt (Diete et al. 2015). However, baits comprising meat are generally more widely used to target quolls (McLean et al. 2015; Austin et al. 2017). Developing a meat-based wax bait or dehydrated sausage may provide a long-life attractant that could be used to target native carnivores in northern Queensland.

The number of camera events across the 21 days of each survey period varied considerably between baits. Sesame oil wax bait appeared to perform fairly consistently, maintaining moderate number of camera events throughout the course of the 21 day period. Standard peanut butter bait and peanut butter wax bait had reduced sampling efficiency towards the end of the survey duration. The reduced sampling efficiency of the peanut-based baits could be due to a decline in bait attractiveness due to bait desiccation and mould growth.

Using wax as a medium to prolong the life of baits in this study was effective as the wax block baits appeared to maintain their integrity far better than the standard peanut butter bait. Sesame oil wax bait appeared to maintain its integrity far better than the other bait types as no mould was evident on any of the sesame oil wax baits at the end of each trial period. Given that peanut butter wax bait was equally as effective as the standard peanut butter bait at luring small native mammals, peanut butter wax block baits may provide an effective long-lasting bait for sampling native mammals in northern Queensland. Further studies are required to determine the effectiveness of adding wax to baits to increase bait longevity up to and beyond the

31 days required to improve the detection of small and medium sized mammals in northern Queensland savanna (chapter 3).

4.4.1 Management implications

This study has demonstrated that peanut butter wax bait is equally as effective as standard peanut butter bait at luring mammals at the MTSWRNR over a 21 day period. The wax formulation appears to have improved bait life, whilst not compromising bait attractiveness. This study further emphasises the suitability of standard peanut butter bait for attracting a wide range of mammal species during short camera trapping studies. Unfortunately, due to time constraints, it was not possible to test the longevity of the baits for more than 21 days. Due to the long camera trap deployment durations required to detect some species in northern Queensland, a comparison between the standard peanut butter bait and peanut butter wax bait needs to be undertaken over an extended time period. Further, developing longer life meat-based baits either as a wax block or dehydrated sausage may assist in improving the detection of carnivorous mammals in ecological surveys.

4.5 CONTRIBUTION TO AUTHORSHIP

The following outlines my contribution to the authorship of this chapter in the thesis:

I made substantial contribution to the conception and design of the project. I collected all data with small amounts of help from volunteers. I received some assistance from Dr. Carly Starr, Dr. Luke Leung and Dr. Allan Lisle for project design and methodologies.

I drafted the entire chapter and developed all the maps and figures. I undertook the statistical analysis with some assistance from Dr. Allan Lisle and Dr. Luke Leung. Interpretation of the research data was undertaken with some assistance from Dr. Luke Leung. I wrote the entire discussion which was then edited by my supervisor Dr. Luke Leung.

CHAPTER 5: GENERAL CONCLUSION

The primary aim of this thesis was to investigate the density of small and medium sized mammals in northern Queensland to establish base-line information for the study of possible species declines. This study focused on three Natural Resource Management regions in northern Queensland: Cape York; Torres Strait; and Northern Gulf.

A single species study was undertaken of the Bramble Cay melomys a small murid rodent located on Bramble Cay, a small isolated (~4ha) sand cay in Torres Strait. The species was previously listed as critically endangered with known history of some decline. Habitat loss due to erosion of the cay and direct mortality from storm surges were implicated as major threats to this species. The study aimed to update the conservation status of the species with an attempt to identify any important factors responsible for any decline and to recover any remaining individuals for a captive insurance population. During three survey periods (December 2011, March 2014 and August–September 2014), a total of 1170 small mammal trap-nights, 60 camera trap-nights, 5 h of nocturnal searches and 5 h of diurnal searches were undertaken on Bramble Cay. All three survey periods failed to detect any Bramble Cay melomys. The island had experienced a recent, severe reduction in vegetation, which is the primary food resource for the Bramble Cay melomys. Herbaceous cover on the cay decreased from 2.16 ha in 2004 to 0.065 ha in March 2014 before recovering somewhat to 0.19 ha in August–September 2014. The results demonstrate that this rodent species has now been extirpated on Bramble Cay. The vegetation decline was probably due to ocean inundation resulting from an increased frequency and intensity of weather events producing extreme high-water levels and storm surges, in turn caused by anthropogenic climate change. The loss of the Bramble Cay melomys from Bramble Cay probably represents the first documented mammalian extinction due to human-induced climate change. Unfortunately, despite there being a documented decline in the species, monitoring of the species was not ongoing between 2004 and 2011, a period when the species became extinct on Bramble Cay. The loss of this species on Bramble Cay highlights the importance of population monitoring and implementation of recovery programs when required to ensure the persistence of small mammals threatened with extinction.

Historical mammal population data was collated for the Northern Gulf region collected using the standardised 1 hectare plot survey methods (430, 1 hectare plot surveys) (Eyre et al. 2012). An additional 63, 1-hectare plot surveys were undertaken across the region to supplement the data. A total of 3,316 cage trap nights; 34,480 Elliot trap nights; 6,270 pitfall trap nights and 298.5 hours of spotlighting were conducted, capturing 461 individuals, from 24 species. Relative abundance was calculated for each species of mammal captured in Northern Gulf region. The relative abundance, species richness and capture rates of small and medium sized mammals identified in the Northern Gulf region is considerably lower than those reported in other studies across the top end of other parts of Northern Australia (Woinarski et al. 2010; Legge et al. 2011; Olds et al. 2016; Olds et al. 2017). An exception to this is the common brushtail possum (Woinarski *et al*. 2010) which was the most common species recorded in the Northern Gulf region, although considered to be potentially declining in northern Queensland (Winter 2007). It is uncertain if mammals have declined across Northern Gulf region (due to a lack of historic sampling in this region) or if these animals are naturally absent or at low abundance and richness.

Higher abundance of small and medium sized mammals were found in Blackbraes National Park and its neighbouring pastoral property in the Einasleigh Uplands bioregion. This area is a rare representation of upland savanna where some mammal species declining elsewhere in northern Australia persist at high abundance (Vanderduys et al. 2012) The persistence of mammals here is perhaps due to the area having low impacts from cattle grazing due to the area containing heart-leaf poison-bush (*Gastrolobium grandiflorum*)**,** a plant toxic to cattle.

When camera traps are integrated into the systematic plot-based survey methods, camera trapping significantly increased the detection of three species of native medium sized terrestrial mammal (northern quoll, northern brown bandicoot and short-beaked echidna) and two invasive species (dingo and feral pig). Of these, the northern quoll and northern brown bandicoot are considered declining in other parts of Australia (Braithwaite & Muller 1997; Woinarski et al. 2001; Pardon et al. 2003).

Eight native mammal species were detected exclusively by camera traps, five of these species were detected on only one plot.

The northern quoll, which is currently listed as endangered (*EPBC Act*) were found to be widely distributed across the experimentally surveyed areas in the Cape York Management Region, including lower elevation habitats where quolls have presumably declined. Camera trap images also showed some evidence of quolls avoiding cane toads suggesting the species has developed prey avoidance strategies and now occur sympatrically with cane toads. The northern brown bandicoot was exclusively captured using camera traps which is surprising given the species is easily captured using cage traps and spotlighting in other studies (Hall 1983; Woinarski et al. 2004; O'Hara 2014). The low captures using the systematic plot-based survey methods was likely due to trap averse behaviour as some of the detections were on the first night of camera trapping.

The standard plot survey methodologies detected the grassland melomys on significantly more plots than camera traps. This result was likely due to a reluctance for the species to venture out from vegetative cover and were mostly predominantly captured in Elliott traps, which are set amongst vegetation. One downfall of using camera traps was the inability for all rodents to be identified to species level.

Camera trapping was particularly effective in detecting introduced species as bycatch (four of the five introduced species were detected only by camera traps), two of these species (dingo and feral pig) were detected significantly more by camera traps. Camera traps are increasingly being used to monitor the abundance of pest animal populations in response to control programs (Cowled et al. 2006; Robley et al. 2008; Robley 2010; Bengsen et al. 2011; Bengsen 2014). Incorporating camera traps into fauna survey methodologies provides useful information on pest species abundance and distribution and may be useful for determining how pest animal populations influence small and medium sized mammal populations.

Many of the species detected using camera traps had highly variable rates until first detection across the plots. Three species, the rakali water rat, rufous bettong and red-cheeked dunnart were detected by cameras only on 12th, 25th and 30th trap nights respectively, suggesting these species may be relatively trap averse, low in abundance or the camera trap was not located in the core home range. This data suggests that increasing camera trap deployment rates may reduce the chance of surveys making a type II error for some species and enhance data for cryptic or rare species.

Two wax block baits were developed with the aim to improve bait longevity, whilst not impacting on bait attractiveness. The effectiveness of the two wax block baits were tested for their attractiveness for camera trapping small and medium sized mammals in tropical savanna and also tested the longevity of the baits over a 21 day period. Standard peanut butter bait performed the best for detecting small native mammal richness. Although there was no statistical difference in the effectiveness of standard peanut butter bait and peanut wax baits for detecting small native mammal richness. Using wax to prolong the life of the bait did not cause any reduction in the attractiveness of bait to small native mammals. However, there also appears to be no benefits of using a wax bait for camera trapping over time periods less than 21 days in duration.

Sesame oil bait performed the best for attracting agile wallabies and using sesame oil wax blocks provide a better longer life bait option for species specific studies. This result contrasted with a similar study undertaken by Diete et al. (2015) where agile wallabies appeared to have no preference for bait type and were equally attracted to a control bait station containing no scent lure. As we did not use an unbaited control in this study and standard peanut butter bait acted as our control, it is possible that the wallabies were attracted to the bait station, rather than the bait.

We did not find preference for bait type for any of the other species in this study. Surprisingly rodents were not detected by standard peanut butter bait, which is likely due to low abundance across the reserve rather than bait preference. Various rodent species were detected using peanut butter bait during our camera trapping surveys in chapter three.

The number of camera events across the 21 days of each survey period varied considerably between baits. Standard peanut butter bait and peanut wax bait had reduced sampling efficiency towards the end of the survey duration which could have been due to a decline in bait attractiveness due to bait desiccation and mould growth.

Using wax as a medium to prolong the life of baits was effective as both sesame oil and peanut wax baits grew considerably less mould than peanut butter bait. Sesame oil appeared to maintain its integrity the best out of the two wax baits as none of the bait blocks grew mould. Given the comparable attractiveness of the peanut wax bait and standard peanut butter bait, the peanut wax bait may provide an effective long-lasting bait for sampling native mammals in northern Queensland. Further studies are required to determine the effectiveness of adding wax to baits to increase bait longevity up to and beyond the 31 days required to improve the detection of small and medium sized mammals in northern Queensland savanna (chapter 3).

5.1 MANAGEMENT IMPLICATIONS

The results of the preceding studies show that populations of small and medium-sized mammals are critically low in some parts of northern Queensland. Urgent actions may be required if declines in small and medium sized mammals are occurring in northern Queensland. The extinction of the Bramble Cay melomys from Bramble Cay highlights the importance of ongoing systematic monitoring programs for threatened species. Perhaps, the extinction of the species could have been avoided if the monitoring program continued past 2004 and a captive insurance population was established sooner in the extinction trajectory.

Studies incorporating camera trapping are required to assess the conservation status of species found in northern Queensland that are known to be declining in other parts of northern Australia (e.g. northern quoll, common brushtail possum, black-footed tree rat, northern brown bandicoot). An ongoing small mammal monitoring program, incorporating camera traps into the recommended state guidelines, should be undertaken, particularly in areas known to have reasonable numbers of small and medium sized mammals (e.g. Blackbraes National park and the surrounding properties, South Endeavour, Kings Plains and Alkoomie Nature refuges). Incorporating camera trapping into the recommended state guidelines may improve our knowledge of the distribution of small and medium sized mammals, particularly for rare and elusive species.

Increasing camera trap deployment times (up to 37 days) would assist in improving the effectiveness of the monitoring programs in detecting small and medium sized mammals. The results of the final experiment suggest the peanut wax bait could offer a robust long-life attractant for use in camera trapping studies where longer camera trap deployment times are required.

5.2 FUTURE STUDIES

Due to the low detection rates of mammals in northern Queensland, the need for improved survey methods is urgently required. Adding camera trapping into the recommended state guidelines for fauna surveys will improve detection rates for many mammal species. The detection of a wider range of species could potentially be improved by increasing the number of camera traps deployed in each plot and using a range of bait types (e.g. incorporating meat-based baits for carnivorous species). Studies investigating camera trap arrays and combinations of baits should be undertaken to determine the most effective number of camera traps and range of baits to attract a wider range of species.

Further research could also investigate ideal camera trap deployment durations for northern Queensland, as some species took up to 37 days to be detected on some plots. Due to the long camera trap deployment durations required to detect some species in northern Queensland, further comparison between the standard peanut butter bait and peanut wax bait is required to take place over an extended time period to determine if peanut wax bait does provide a longer-life alternative to standard peanut butter bait. Furthermore, developing longer life meat-based baits either as a wax block or dehydrated sausage may assist in improving the detection of carnivorous mammals in ecological surveys.

5.3 CONCLUSION

This study has produced four main outcomes: an improved understanding of the conservation status of the Bramble Cay melomys, which was Queensland's most threatened mammal; an improved understanding of the abundance of mammals in the Northern Gulf region; understanding of the benefits of integrating camera traps into the standard plot survey methodologies for detecting small and medium sized mammals; and identification of a wax bait that could potentially replace standard peanut butter bait for camera trapping surveys where long deployment durations are required.

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APPENDIX 1

THE UNIVERSITY OF QUEENSLAND

UQ Research and Innovation
Director, Research Management Office
Nicole Thompson

ANIMAL ETHICS APPROVAL CERTIFICATE 02-Oct-2013

Activity Details

Chief Investigator:	Dr Luke Leung, Agriculture and Food Sciences
Title:	Terrestrial vertebrate fauna surveys in Cape York
AEC Approval Number:	SAFS/095/13
Previous AEC Number:	
Approval Duration:	03-Oct-2013 to 03-Oct-2016
Funding Body:	Aust. Post Grad. Award Scholarship
Group:	Native and exotic wildlife and marine animals
Other Staff/Students:	Natalie Waller, Juli Broken-Brow, Tyrone Lavery
Location(s):	Other Queensland Location

Summary

Subspecies	Strain	Class	Gender	Source	Approved	Remaining
Amphibians	Marbled Frog (Limnodynastes convexiusculus)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Ornate Burrowing Frog (Limnodynates ornatus)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Scarlet-sided pobblebonk (Limnodynastes terraereginae)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Northern sedgefrog (Litoria bicolor)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Green treefrog (Litoria caerulea)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Graceful tree frog (Litoria gracilenta)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Striped Rocket Frog (Litoria nasuta)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Tawny rocket frog (Litoria nigrofrenata)	Adults	Unknown	Natural Habitat	200	200

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Amphibians	Naked treefrog (Litoria rubella)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Red-eye Green treefrog	Adults	Unknown	Natural Habitat	200	200
Amphibians	Shrill chirper (Sphenophryne gracilipes)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Australian bullfrog (Rana daemeli)	Adults	Unknown	Natural Habitat	200	200
Amphibians	Cane Toad (Bufo marinus)	Adults	Unknown	Natural Habitat	200	200
Dasyurids	Chestnut dunnart (Sminthopsis archeri)	Adults	Unknown	Natural Habitat	200	200
Dasyurids	Brush-tailed phascogale (Phascogale tapoatafa)	Adults	Unknown	Natural Habitat	200	200
Dasyurids	Northern Quoll (Dasyurus hallucatus)	Adults	Unknown	Natural Habitat	200	200
Dasyurids	Cinnamon Antechinus (Antechinus leo)	Adults	Unknown	Natural Habitat	200	200
Dasyurids	Red-cheeked Dunnart (Sminthopsis virginiae)	Adults	Unknown	Natural Habitat	200	200
Dasyurids	Various Bandicoots	Adults	Unknown	Natural Habitat	200	200
Lizards	Gecko (Gehyra dubia)	Adults	Unknown	Natural Habitat	200	200
Lizards	Asian House Gecko (Hemidactylus Frenatus)	Adults	Unknown	Natural Habitat	200	200
Lizards	Mourning gecko (Lepidodactylus lugubris)	Adults	Unknown	Natural Habitat	200	200
Lizards	Pelagic Gecko	Adults	Unknown	Natural Habitat	200	200
Lizards	Zig-zag gecko (Oedura rhombifer)	Adults	Unknown	Natural Habitat	200	200
Lizards	Giant tree gecko (Rhacodactylus australis)	Adults	Unknown	Natural Habitat	200	200
Lizards	Burtons Legless Lizard (Lialis burtonis)	Adults	Unknown	Natural Habitat	200	200
Lizards	Frillneck Lizard (Chlamydosaurus kingii)	Adults	Unknown	Natural Habitat	200	200

Lizards	Two-lined dragon (Diporiphora bilineata)	Adults	Unknown	Natural Habitat	200	200
Lizards	Gould's (Sand) monitor	Adults	Unknown	Natural Habitat	200	200
Lizards	Mangrove monitor	Adults	Unknown	Natural Habitat	200	200
Lizards	Spotted tree monitor (Varanus timorensis)	Adults	Unknown	Natural Habitat	200	200
Lizards	Carlia longipes	Adults	Unknown	Natural Habitat	200	200
Lizards	Carlia storri	Adults	Unknown	Natural Habitat	200	200
Lizards	Cryptoplepharus virgatus	Adults	Unknown	Natural Habitat	200	200
Lizards	Cryptoplepharus litoralis	Adults	Unknown	Natural Habitat	200	200
Lizards	Ctenotus spaldingi	Adults	Unknown	Natural Habitat	200	200
Lizards	Major Skink (Egernia frerei)	Adults	Unknown	Natural Habitat	200	200
Lizards	Glaphyromorphus crassicaudus	Adults	Unknown	Natural Habitat	200	200
Lizards	Glaphyromorphus nigricaudis	Adults	Unknown	Natural Habitat	200	200
Lizards	Lygisaurus sesbrauna	Adults	Unknown	Natural Habitat	200	200
Lizards	Lygisaurus macfarlani	Adults	Unknown	Natural Habitat	200	200
Lizards	Eastern Blue Tongue Skink	Adults	Unknown	Natural Habitat	200	200
Lizards	Blind snake (Ramphotyphlops polygrammicus)	Adults	Unknown	Natural Habitat	200	200
Monotremes	Echidna (Tachyglossus aculeatus)	Adults	Unknown	Natural Habitat	200	200
Native Rats and Mice	Water Rat	Adults	Unknown	Natural Habitat	200	200
Native Rats and Mice	White-tailed Rat	Adults	Unknown	Natural Habitat	200	200
Native Rats and Mice	Melomys (M capensis and M burtoni)	Adults	Unknown	Natural Habitat	200	200
Native Rats and Mice	Cape York Rat (Rattus leucopus)	Adults	Unknown	Natural Habitat	200	200
Native Rats and Mice	Cane field Rat (Rattus sordidus)	Adults	Unknown	Natural Habitat	200	200
Native Rats and Mice	Delicate Mouse (Pseudomys delicatus)	Adults	Unknown	Natural Habitat	200	200
Other Birds	Various Bird Species for observation only	Adults	Unknown	Natural Habitat	200	200

Possums and Gliders	Common Brush Tail Possum (Trichosurus vulpecula)	Adults	Unknown	Natural Habitat	200	200
Possums and Gliders	Sugar Glider (Petaurus breviceps)	Adults	Unknown	Natural Habitat	200	200
Possums and Gliders	Striped possum (Dactylopsila trivirgata)	Adults	Unknown	Natural Habitat	200	200
Possums and Gliders	Grey cuscus (Phalanger orientalis)	Adults	Unknown	Natural Habitat	200	200
Possums and Gliders	Spotted cuscus (Spilocuscus maculatus)	Adults	Unknown	Natural Habitat	200	200
Possums and Gliders	Feather Tail Glider (Acrobates pygmaeus)	Adults	Unknown	Natural Habitat	200	200
Snakes	Various species (Observation only)	Adults	Unknown	Natural Habitat	200	200

Permit(s):

Scientific Purposes Permit WITK13159813	09-Sep-2013 to 08-Sep-2016
Scientific Purposes Permit WISP13159613	09-Sep-2013 to 08-Sep-2016

Proviso(s):

<u>Approval Details</u>

Description	Amount	Balance
Amphibians (Australian bullfrog (Rana daemeli), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Cane Toad (Bufo marinus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Graceful tree frog (Litoria gracilenta), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Green treefrog (Litoria caerulea), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Marbled Frog (Limnodynastes convexiusculus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Naked treefrog (Litoria rubella), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Northern sedgefrog (Litoria bicolor), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Ornate Burrowing Frog (Limnodynates ornatus), Unknown, Adults, Natural Habitat)		

17 May 2013 Initial approval	200	200
Amphibians (Red-eye Green treefrog, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Scarlet-sided pobblebonk (Limnodynastes terraereginae), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Shrill chirper (Sphenophryne gracilipes), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Striped Rocket Frog (Litoria nasuta), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Amphibians (Tawny rocket frog (Litoria nigrofrenata), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Dasyurids (Brush-tailed phascogale (Phascogale tapoatafa), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Dasyurids (Chestnut dunnart (Sminthopsis archeri), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Dasyurids (Cinnamon Antechinus (Antechinus leo), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Dasyurids (Northern Quoll (Dasyurus hallucatus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Dasyurids (Red-cheeked Dunnart (Sminthopsis virginiae), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Dasyurids (Various Bandicoots, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Asian House Gecko (Hemidactylus Frenatus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Blind snake (Ramphotyphlops polygrammicus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Burtons Legless Lizard (Lialis burtonis), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Carlia longipes, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Carlia storri, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Cryptoplepharus litoralis, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Cryptoplepharus virgatus, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Ctenotus spaldingi, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200

Lizards (Eastern Blue Tongue Skink, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Frillneck Lizard (Chlamydosaurus kingii), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Gecko (Gehyra dubia), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Giant tree gecko (Rhacodactylus australis), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Glaphyromorphus crassicaudus, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Glaphyromorphus nigricaudis, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Gould's (Sand) monitor, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Lygisaurus macfarlani, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Lygisaurus sesbrauna, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Major Skink (Egernia frerei), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Mangrove monitor, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Mourning gecko (Lepidodactylus lugubris), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Pelagic Gecko, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Spotted tree monitor (Varanus timorensis), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Two-lined dragon (Diporiphora bilineata), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Lizards (Zig-zag gecko (Oedura rhombifer), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Monotremes (Echidna (Tachyglossus aculeatus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Native Rats and Mice (Cane field Rat (Rattus sordidus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Native Rats and Mice (Cape York Rat (Rattus leucopus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Native Rats and Mice (Delicate Mouse (Pseudomys delicatus), Unknown, Adults, Natural Habitat)		

17 May 2013 Initial approval	200	200
Native Rats and Mice (Melomys (M capensis and M burtoni), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Native Rats and Mice (Water Rat, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Native Rats and Mice (White-tailed Rat, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Other Birds (Various Bird Species for observation only, Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Possums and Gliders (Common Brush Tail Possum (Trichosurus vulpecula), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Possums and Gliders (Feather Tail Glider (Acrobates pygmaeus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Possums and Gliders (Grey cuscus (Phalanger orientalis), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Possums and Gliders (Spotted cuscus (Spilocuscus maculatus), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Possums and Gliders (Striped possum (Dactylopsila trivirgata), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Possums and Gliders (Sugar Glider (Petaurus breviceps), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200
Snakes (Various species (Observation only), Unknown, Adults, Natural Habitat)		
17 May 2013 Initial approval	200	200

Please note the animal numbers supplied on this certificate are the total allocated for the approval duration

Please use this Approval Number:

1. When ordering animals from Animal Breeding Houses
2. For labelling of all animal cages or holding areas. In addition please include on the label, Chief Investigator's name and contact phone number.
3. When you need to communicate with this office about the project.

It is a condition of this approval that all project animal details be made available to Animal House OIC.
(UAEC Ruling 14/12/2001)

The Chief Investigator takes responsibility for ensuring all legislative, regulatory and compliance objectives are satified for this project.

This certificate supercedes all preceeding certificates for this project (i.e. those certificates dated before 02-Oct-2013)

APPENDIX 2

THE UNIVERSITY OF QUEENSLAND

PLEASE KEEP THIS FORM IT IS YOUR RECORD OF YOUR AEC APPROVAL NUMBER

Ms Ann Higgins
Animal Welfare Coordinator
Research and Research Training Division
Cumbrae Stewart Building (72)
St Lucia Q 4072
Ph: (07) 3365 2713 Fax: (07) 3365 4455
Email: a.higgins@research.uq.edu.au

ANIMAL ETHICS APPROVAL CERTIFICATE

Date: 18-Oct-2010

Dear Dr Luke Leung, Animal Studies

The following project: *The abundance, distribution and habitat use of the Bramble Cay Melomys, Roof Rat and the House Mouse on Bramble Cay and two nearby islands in Torres Strait*

Requesting funding from (Grant Awarding Body):- involves animal experimentation. It has been reviewed and ethical clearance obtained from the University Animal Ethics Committee (Native and exotic wildlife and marine animals).

AEC Approval Number: SAS/229/10

Previous AEC Number:

Approval Duration: 18-Oct-2010 to 18-Oct-2013

Permit(s): Scientific Purposes Permit WISP07911010 01-Sep-2010 to 14-Aug-2015

SUBSPECIES	STRAIN	CLASS	GENDER	SOURCE	AMOUNT
Native Rats and Mice	Bramble Cay Melomys	Other	Mix	Natural Habitat	600
Native Rats and Mice	Black Rat (Rattus rattus)	Other	Mix	Natural Habitat	600
Native Rats and Mice	House Mouse (Mus musculus)	Other	Mix	Natural Habitat	600

Proviso(s):

Please use this Approval Number:

1. When ordering animals from Animal Breeding Houses
2. For labelling of all animal cages or holding areas. In addition please include on the label, Chief Investigator's name and contact phone number.
3. When you need to communicate with this office about the project.

It is a condition of this approval that all animal usage details be made available to Animal House OIC. (UAEC Ruling 14/12/2001)

This certificate supercedes all preceeding certificates for this project (i.e. those certificates dated before 18-Oct-2010

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APPENDIX 3

Photographs of the north-western end of Bramble Cay taken from the lighthouse: a) between October 1979 and March 1980 (David Carter); b) December 2012 (Stan Lui); c) March 2014 (Ian Gynther); and August–September 2014 (Ian Gynther).

a)

b)

c)

d)

APPENDIX 4

Photographs taken from various locations on Bramble Cay in August–September 2014 demonstrating: a) flattened, dead vegetation near the southern shoreline; b) dead or badly damaged vegetation in low-lying areas parallel to the northern shoreline; c) salt-affected vegetation in a swale or depression, with the black arrow demonstrating the assumed direction of seawater flow from south-east to north-west, as determined by the pattern of deposition of beach-washed items and other material; and d) accumulations of driftwood, dead bird eggs and other debris present at the north-western end of the swale (all photos, Ian Gynther).

a)

b)

c)

d)

APPENDIX 5

Observations of northern quolls actively avoiding toads. a. Northern quoll approaches toad at bait station; b. Northern approaches toad at bait station a second time; c. northern quoll retreats from toad. Arrow indicates location of toad.

a)

b)

c)

APPENDIX 6

List of mammal species known to occur at the Mareeba Tropical Savanna and Wetland Reserve Nature Refuge and species detected during this survey.

Scientific name	Common name	Detected 2016
Aepyprymnus rufescens	Rufous bettong	**X**
Dasyurus hallucatus	Northern quoll	**X**
Hydromys chrysogaster	Water rat	
Isoodon macrourus	Northern brown bandicoot	**X**
Notamacropus agilis	Agile wallaby	**X**
Macropus antilopinus	Antilopine wallaroo	
Macropus giganteus	Eastern grey kangaroo	**X**
Macropus parryi	Whiptail wallaby	
Macropus robustus	Common wallaroo	**X**
Melomys burtoni	Grassland melomys	
Mesembriomys gouldii	Black-footed tree-rat	
Petauroides Volans	Greater glider	
Petaurus breviceps	Sugar glider	
Petaurus norfolcensis	Squirrel glider	
Petrogale assimilis	Allied rock-wallaby	
Petrogale godmani	Godman’s rock-wallaby	
Petrogale Mareeba	Mareeba rock-wallaby	
Phascogale tapoatafa	Brush-tailed phascogale	
Pseudocheirus peregrinus	Common ringtail possum	
Rattus sordidus	Canefield rat	
Tachyglossus aculeatus	Short-beaked echidna	**X**
Trichosurus vulpecula	Common brushtail possum	**X**