Northern Australia Environmental Resources Hub

National Environmental Science Programme

Experimental evaluation of the role of feral cat predation in the decline of small mammals in Kakadu National Park

Final report

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NORTHERN TERRITORY GOVERNMENT

kakadu NATIONAL PARK

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Cover photographs
Front cover: Feral cat captured on motion detection camera (photo DENR).
Back cover: Predator exclusion fence (photo NAER Hub).

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EXECUTIVE SUMMARY

Small and medium-sized native mammals have suffered severe declines in tropical savannas across northern Australia, including within protected areas such as Kakadu National Park. Several factors have been implicated in these declines but predation by feral cats, *Felis catus*, has been identified as potentially the most direct cause of decline for many species. Predator-proof fences have previously been used successfully to exclude feral cats from targeted areas for threatened species conservation. In order to evaluate the role of cat predation in mammal declines, this project used exclosures to experimentally evaluate vertebrate population responses to cat-exclusion in Kakadu National Park. Additionally, the project increased knowledge of the ecology of feral cats and dingoes in Kakadu National Park.

Two 64 ha predator-proof exclosures were constructed at Kapalga, Kakadu National Park in December 2013. Four non-fenced sites of equal size were established to serve as controls. Baseline population data of small- and medium-sized mammals, and reptiles, was obtained prior to fence completion. Following fence construction; vertebrate sampling was undertaken by trapping three times per year over two years, from November 2013 to December 2015. Cats were detected using camera traps and one cat was tracked using a GPS collar. The diets of cats and dingoes were evaluated from scats collected in the study area.

No measurable population responses of small-mammals were detected in response to cat-exclusion due to low capture rates throughout the experimental area. Northern Quolls were captured relatively frequently in the unfenced study sites. In contrast, reptile abundance doubled over two years in the fenced exclosures but remained unchanged in the unfenced sites. Additionally, reptile abundance increased at sites with time since fire regardless of treatment. Reptile species richness increased inside fenced areas; however, large seasonal variation in captures of species weakened the significance of this result.

A cat density estimate of 0.19 individual cats per km^2 (~ 1 cat per 5 km^2) was calculated for Kapalga, consistent with estimates obtained from other areas in northern Australia. The collared cat used approximately 869 hectares over the 57 days it was tracked. Scat analysis indicated there was some overlap in the diets of dingoes and cats with both species preying on medium-sized mammals such as Northern Brown Bandicoots and Brushtail Possums. Cats predominately preyed upon mammals which made up almost 75 % of their diet, but also preyed on a diversity of small-sized animals including birds and reptiles. Dingoes predominately preyed upon macropods.

This experiment was inconclusive for impacts of cats on small mammals, possibly due to short duration of the study limiting detection of measureable rates of increase. Many of the mammal species which would be expected to be cat-susceptible still persist in the landscape but at low densities, but predators may be impeding recolonization of fenced exclosures.

Northern Quolls are breeding in the Kapalga area, making this an important population in Kakadu, and warrants proactive monitoring and evaluation. The cat exclusion experiment demonstrated that predation by feral cats is exerting considerable pressure on small reptile diversity in savanna woodland habitat. Fire is also exerting suppressive effects on small reptile populations. Further monitoring of reptiles at these experimental plots would yield important information on the relative vulnerability of different components of the reptile community, and help to hone management priorities.

1 INTRODUCTION

Small and medium-sized native mammals have suffered severe declines across northern Australia (Ziembicki *et al.* 2015), including within protected areas such as Kakadu National Park (Woinarski *et al.* 2010). Several factors have been implicated in the decline including changes in habitat structure and productivity resulting from altered fire regimes and the introduction of livestock, poisoning from cane toads, and emergent wildlife disease (Woinarski *et al.* 2011). However, predation by feral cats, *Felis catus*, has been identified as potentially the major proximal cause of these declines (Woinarski *et al.* 2015).

Cats were introduced to Australia by European colonists and are now ubiquitous on mainland Australia and many off-shore islands, and have been implicated in the decline and extinction of many mammal species (Burbidge and Manly 2002; Medina *et al.* 2013; Frank *et al.* 2014). Cats are a generalist predator that prey on a wide range of species and are capable of switching prey as preferred prey are depleted (Doherty *et al.* 2015b). Due to their body size cats can access refugia used by small and medium-sized mammals, which make up a large portion of the feral cat diet (Kutt 2012; Doherty *et al.* 2015b). Even when cats are at low densities, they can deplete local populations of small and medium-sized mammals, increasing extinction risk of threatened species (Moseby *et al.* 2015).

Until recently empirical data on feral cat ecology and demography in Northern Australia were limited; and evidence for evaluating the direct impact of predation by cats on small and medium-sized mammal populations (and other vertebrates) in northern Australian savannas has also been lacking. This has hampered the evaluation of their role, relative to other environmental factors, in the observed mammal declines of northern Australia.

In a recent study, Frank *et al.* (2014) compared the survival of translocated Long-haired Rats, *Rattus villosissimus*, in enclosures that were either accessible or inaccessible to feral cats. Rats in the accessible enclosures were rapidly extirpated, whilst rats in the cat-proof enclosures persisted. However, the fitness of the rats used in this experiment may have been compromised, as they were sourced from a cat-free island and greater than 90% were captive-bred prior to release, limiting general inference from the study. Tuft *et al.* (2014) subsequently replicated the study with wild translocated Pale Field-rats, *Rattus tunneyi*, from an area where they coexisted with feral cats. Again, the rat populations in the cat-accessible plots were rapidly extirpated. These findings suggest that small mammal populations may be difficult to re-establish in areas where feral cats are present.

McGregor *et al.* (2014) found that cats preferentially hunt in areas with little ground-layer vegetation cover, especially heavily grazed areas and areas burnt by high intensity fires, and particularly if those habitats had a high abundance of small mammals. Furthermore, the hunting efficiency of cats was significantly increased in areas in open habitats (McGregor *et al*. 2015a), indicating that habitat changes associated with high intensity fires and livestock grazing may amplify feral cat predatory impacts on native mammal populations.

Effective broad scale management of feral cats on mainland Australia is currently unfeasible due to a lack of cost-effective methods. In comparison, predator-proof fencing has been used successfully to conserve threatened species in several countries, including Australia, to protect native species from exotic predators. Whilst as a management tool predator-proof fencing has local benefits, equally there are both financial and ecological costs (reviewed in

Hayward and Kerley 2009). However, as an experimental tool predator-proof fences provide an opportunity to investigate population responses of small vertebrates *in-situ* when predatory pressure from exotic predators is removed. Significant work has been undertaken in Australia to optimise fence designs to successfully exclude feral cats (Moseby and Read 2006; Robley *et al.* 2007) making it feasible to effectively exclude cats from targeted areas.

In order to assess the contribution of predation by feral cats to the observed mammal declines, it is necessary to quantify their predatory impact on natural populations. Furthermore, an understanding of the impact of feral cats relative to, or in synergy with, other potential drivers of mammal decline is desirable in order to make informed decisions about where and how to best direct management interventions. This project sought to improve understanding of the ecology of feral cats in savanna woodlands in order to inform management options for addressing the observed mammal declines. Specifically we evaluated the impact of feral cat predation on small vertebrate populations in Kakadu National Park, northern Australia, and the ability of populations to recover after cat exclusion.

1.1 Project Objectives

The primary objective of this project was to experimentally evaluate the population responses of small and medium-sized native mammals to the exclusion of feral cats. Secondary objectives were to:

1) evaluate the population responses of reptiles to exclusion of feral cats;
2) examine potentially interactive effects on mammal and reptile diversity of fire and cat exclusion;
3) increase knowledge of the ecology of feral cats and dingoes in Kakadu National Park, through
    a) comparative examination of diets of feral cats and dingoes;
    b) trial and evaluation of methods for estimating population density of cats and dingoes; and
    c) trialling of trapping methods for feral cats.

2 STUDY SITE SELECTION, EXCLOSURE CONSTRUCTION AND MANAGEMENT

2.1 Site Selection

The target habitat for implementation of the experiment was in lowland open forest, a dominant habitat occurring almost continuously across higher-rainfall areas of northern Australia. Lowland habitats have historically supported a rich and diverse mammal fauna but marked declines have occurred in the distribution and abundance of this mammal fauna in recent history (Woinarski *et al.* 2001). A desk top investigation was undertaken using Google Earth, topographic maps, available vegetation mapping (Schodde *et al.* 1986), fire history maps (MODIS fire scars scale 250 m x 250 m), Northern Territory Government Flora and Fauna Atlas data and local knowledge, to identify potentially suitable locations to carry out the project. Potential localities were assessed based upon the following criteria, of which some were assessed remotely, and others by on-ground survey.

- Homogeneity of lowland open forest, dominated by *Eucalypus miniata* and *E. tetrodonta*
- Flat topography extensive enough to construct and maintain integrity of 1 km^2 predator-proof fences
- Year-round accessibility for sampling and maintenance purposes
- Presence of residual small mammal populations in the area

2.2 Consultation Process

Initially the intention was to replicate the cat exclusion experiment at two geographically separate locations in the Top End. Kakadu National Park (KNP) was the primary location with Litchfield National Park, Garig Gunak Barlu (Cobourg) National Park or Bathurst Island considered for secondary locations.

Litchfield National Park was rejected as a secondary site due to on-going land ownership disputes. Preliminary surveys undertaken at sites within Litchfield National Park also returned few mammal captures. Potential sites within Garig Gunak Barlu National Park were identified through consultation with Park staff, GIS and a site visit. A consultation booklet was created for the traditional owners and a presentation given to the board by DENR staff. Traditional owners were not convinced that cats were present in Garig Gunak Barlu. Traditional owners requested preliminary surveys be undertaken to confirm the presence of cats within different areas of the park prior to project consideration. This work did not fit into the project time-frames and for this reason Garig Gunak Barlu National Park was rejected as a site for the study.

After discussions with some Tiwi Island traditional owners it became apparent that they had concerns about the impact of cats on small mammals. A proposal to undertake the project on Bathurst Island was presented to the Tiwi Land Council and subsequently supported. Potentially suitable sites were identified on Bathurst Island, and preliminary mammal surveys were undertaken, which returned high numbers of medium-sized mammals (Brushtail Possum *Trichosurus vulpecula,* and Northern Brown Bandicoot *Isoodon macrourus*) but no small-sized mammals, such as Brush-tailed Rabbit-rats *Conilurus penicillatus* (Vulnerable, *EPBC* Act) or other native rodent species. However, the costs of fence construction on the island and logistical challenges prevented proceeding at this location. Consequently efforts were focussed on a single study area in KNP.

The consultation process with traditional owners of KNP was undertaken by KNP management staff with participation by Northern Territory Department of Environment and Natural Resources (DENR) staff. A photo booklet was developed by DENR to aid the consultation process, particularly for people for whom English was not a first language. At times consultation was conducted with an interpreter or a relative without direct responsibility for the land with whom the landowner was comfortable. DENR staff gave a presentation to the *Binninj* board of KNP, a management board comprised only of traditional owners. Approval was subsequently granted to construct the experimental exclosures and undertake the research in the Kapalga area of KNP (Figure 1). Under the provisions of the *Environment Protection and Biodiversity Conservation Act 1999*, and outlined in the KNP Plan of Management, an environmental impact assessment was required, and thus undertaken prior to fence construction works. A formal agreement was entered into between Parks Australia and Charles Darwin University for the construction and management of the infrastructure, and relevant Animal Ethics, Northern Territory and Parks Australia research permits were secured for the project.

2.3 Study Design

Due to landscape and resource constraints, the size of the exclosures was reduced from 1 km^2 to 800 x 800 m (64 ha). Six sites were established in the Kapalga area of KNP with similar habitat and landscape characteristics, and comparable fire histories over the previous 10 years. Three different treatments were applied to the six sites to compare and contrast relative responses of small mammal abundance and species composition to treatment effects, whilst controlling for the possible confounding effects of fire. The sites were designed as follows:

- Two fenced sites with 8 m wide perimeter firebreaks with fire suppression undertaken. Fences excluded cats and dogs, medium-sized and large mammals, and introduced livestock, but were permeable to small mammals, reptiles and amphibians (including Cane Toads *Rhinella marina*).
- Two unfenced sites with 8 m wide perimeter firebreaks to prevent fire, and fire suppression undertaken.
- Two unfenced control sites without perimeter firebreaks and no direct fire suppression. These sites were subject to ambient fire regimes in the Kapalga area, where the average fire-return interval is one to two years (NT Infonet, http://www.infonet.org.au).

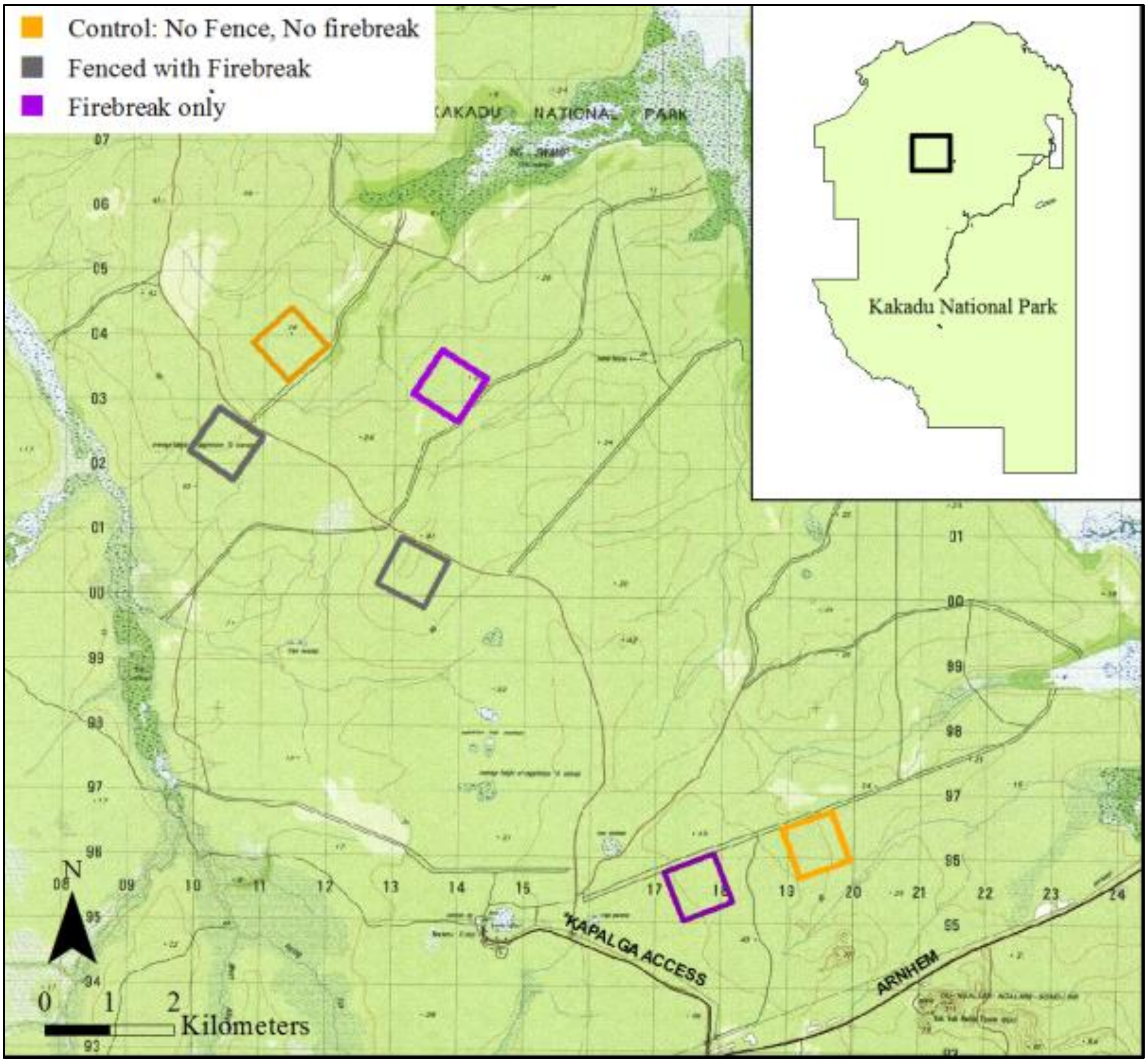


Figure 1. Location of the study site in Kakadu National Park, illustrating geographic locations of experimental sites. Grey squares – fenced sites; Orange squares – unfenced fire-protected sites; Orange squares - unfenced fire-unprotected sites.

2.4 Exclosure Fence Design and Construction

The predator-proof fence was based on a design developed and trialled at Arid Recovery in South Australia to exclude feral cats, foxes and rabbits (Moseby and Read 2006). The fences were 1800 mm tall with a curved floppy section extending 450 mm outward from the top of the fence and internal and external foot aprons extending 550 mm from the base of the fence (Figure 2). The design of the 'floppy-top' is integral to the exclusion of cats from the area of interest by preventing them from gaining purchase on top of the fence. Internal and external foot aprons were constructed to prevent animals from digging under the fence. The key construction features of the fence were as follows:

- The body of fence was constructed from prefabricated 40 mm hexagonal mesh, 'hexmesh'. An 1800 mm wide piece of hexmesh was used on the bottom half of the fence and formed the external foot apron. A 1200 mm wide piece of hexmesh was used on the top half of the fence and formed the floppy top. The two pieces of hexmesh were joined with a single 50 mm overlap at 1250 mm above the ground. The join was secured with fence clips every 10 cm along the length of the fence.

- The internal foot apron was joined to the body of the fence at 300 mm above the ground.
- The section comprising the floppy top was 600 mm long and shaped with 4 mm high tensile supporting wires so that it extended 450 mm outward from the top of the fence. These support wires were hand-bent once threaded through the mesh to form the floppy top arc.
- Straining posts were placed every 100 m with star pickets in between at every 8 m. A total of seven strainer wires were run horizontally up the body of the fence every 300 mm.
- Corners were constructed using galvanised straining posts with an internal angled strut ~ 15 mm from the hexmesh to prevent cats from gaining purchase.
- A single 3 m wide gate was installed at one corner of each fenced site, also constructed with a floppy top to preclude cats.

A fire-break of 8 m (4 m either side of the fence) was cleared prior to fence construction. The fire-break was deliberately minimised to reduce the extent of habitat disturbance. Firebreak clearing and fence construction were put out for tender in early 2013. Land-clearing for the firebreaks was completed at the beginning of September 2013 and fence construction commenced at the end of that month. Construction of both fences was completed in December 2013. An arborist was contracted to remove individual trees and tree-limbs that were considered hazardous to the integrity of the fences.

The fence design utilised 40 mm hexmesh to allow free movement of small-sized mammals, such as native rodents, in and out of the exclosures. The fences were likely impervious to the entry of adults of most medium-sized mammal species, but not juveniles, and all species excluding the Northern Brown Bandicoot would likely be able to climb out of the exclosures. Medium-sized mammals of a highly arboreal nature, such as Black-footed Tree-rats *Mesembriomys gouldii* (Vulnerable, *EPBC* Act), were likely to be able to freely climb both in and out of the exclosures.

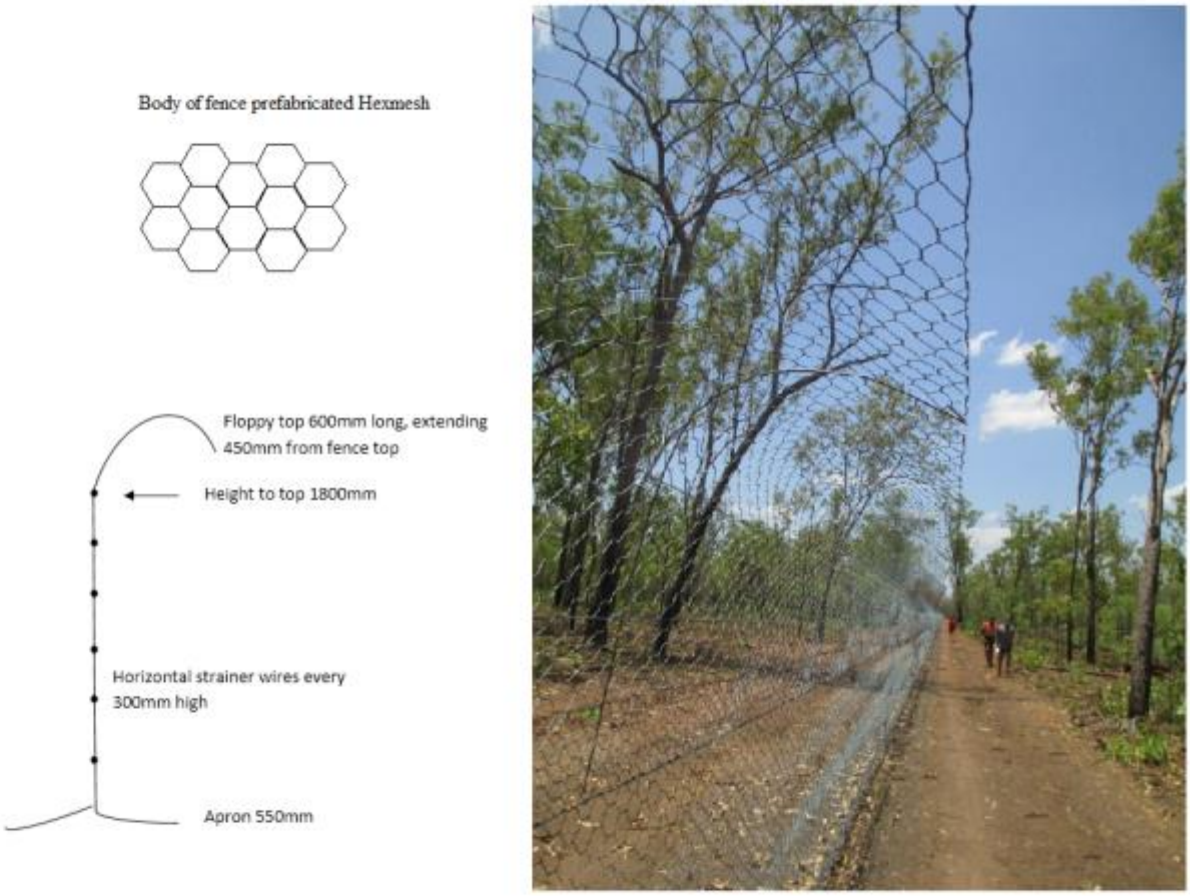


Figure 2. Design of predator-proof fence (left) and image along fence constructed at Kapalga, Kakadu National Park (right).

2.5 Fence and Site Maintenance

Fence and site maintenance were sub-contracted to a local indigenous contractor from KNP. The contractor was responsible for undertaking a visual inspection of the fences on a weekly basis and repairing any damage incurred by animals, heavy rain and fallen trees/branches. Vegetation control by slashing and herbicide application on the fire breaks was undertaken to prevent damage to the fences, to maintain effective fire breaks, and prevent incursion of fire.

Fire management was challenging with all sites burning at some point during the project (Table 1). Prior to fence construction, between June and August 2013, both fence sites and one fire break site were partially burnt. Between October and November 2013, the fenced sites were (unintentionally) burnt again, and both control sites were completely burnt. District rangers from KNP undertook planned back-burning activities around the research sites in the early dry season of 2014. Further back-burning was undertaken later in the dry season of 2014 around fenced and firebreak sites to mitigate spread from approaching fires. Back-burning was undertaken again in May 2015 when one fenced site was threatened with fire, and June 2015 when a firebreak site was threatened. Despite these actions in August 2015 one of the fenced sites was completely burnt (Figure 3). In November 2015, while undertaking vertebrate sampling, a lightning strike ignited a fire on one of the firebreak sites (Figure 4).

Fences were damaged by fallen trees and/or large branches on three occasions between February 2014 and March 2015 (Figure 5). Fighting buffalo caused damage to fences at one site in January 2015 and again in October 2015. In all instances, the fences were repaired as soon as practical. No predator incursions were detected following any of these incidents despite the integrity of the fences being temporarily compromised.

Table 1. Number of times each year of the study that each site was burnt.

Site	Treatment	2013	2014	2015
KAP01	Control	1	0	1
KAP02	Fence	2	0	0
KAP03	Fence	2	0	1
KAP04	Firebreak	1	0	0
KAP05	Firebreak	0	0	1
KAP06	Control	1	0	1

Figure 3. Exclosure following unintentional burn (photo Danielle Stokeld).

Figure 4. Fire at one of the experimental sites captured on camera trap.

Figure 5. Fence damage caused by a fallen tree (photo Khan Spokes)

2.6 Fence Security Monitoring with Camera Traps

In order to confirm that the fenced sites successfully excluded cats, eight camera traps (Reconyx HC550 and HC600) were spaced 400 m apart along the predator fencing in each fenced exclosure. Cameras were mounted on fence pickets approximately 100 cm above the ground. Cameras were oriented to take photos of animals passing in front of the camera on both the inside and outside of each fenced area. Cameras were set to take three photos per trigger with a 1 second interval between photos and no delay between triggers. These cameras were also used to assess presence of other animals such as macropods stranded in the fenced areas.

One male cat was detected on camera inside one enclosure shortly after fence construction was completed. After removal of the cat from within the exclosure (see Section 4.1.1), the two exclosures remained free of cats and dingos throughout the remainder of the study (Table 2). Between November 2013 and December 2015, camera traps captured at least two individual cats walking outside and adjacent to the fenced exclosures. There were 47 detections of cats and 88 detections of dingos outside one exclosure, and 161 cat detections and 147 dingo detections outside the other exclosure.

Numerous other mammal species were also detected on camera traps outside and adjacent to the exclosures including: Buffalo (*Bubalus bubalis*) (Figure 6), Horse *Equus caballus*, Pig *Sus scrofa*, Antilopine Wallaroo *Macropus antilopinus*, Common Wallaroo *Macropus robustus*, Agile Wallaby *Macropus agili*), Echidna *Tachyglossus aculeatus*, Northern Quoll *Dasyurus hallucatus* (Endangered, *EPBC* Act), Northern Brown Bandicoot, and Black-footed Tree-rat (Figure 7).

Figure 6. Buffalo captured on camera trap outside one of the fenced exclosures.

2.7 Macropod Management

Site walk-throughs using 50 – 80 people (i.e. average distance apart of 10 -15 m) were organised to flush any large animals, particularly macropods, from the fenced areas on the final day of fence construction. These walk-throughs varied in success, with most macropods flushed from one fenced site but very few flushed from the other. Two additional walk-throughs were organised in subsequent months but were unsuccessful in removing any additional macropods. Based upon camera trap photos and direct observation it was estimated that 3 Agile Wallaby were stranded in one exclosure and 12 -15 Agile Wallaby in the other.

During the dry season of 2014 and 2015 watering stations were established inside the exclosures adjacent to the gates for animal welfare reasons and to encourage the wallabies to congregate in one area for potential mustering and removal. Wallabies were observed using the watering stations, but use was intermittent and wallabies could not be mustered. Parks staff attempted ground shooting of wallabies, without success. The Kakadu Integrated Feral Team (IFT) subsequently undertook an aerial shoot of the wallabies within the two exclosures in December 2014 and February 2015.

The first aerial cull undertaken by IFT removed two wallabies from one, and eight from the other exclosure. A second aerial cull removed a further one and five wallabies from the two exclosures respectively. However, the ongoing presence of wallabies within the enclosures was confirmed using camera traps. In April 2015, one debilitated wallaby was hand-captured and released outside one exclosure. Despite ongoing attempts at removal, the difficulty involved in culling wallabies in such an overgrown compound meant that at the completion of the project one wallaby remained in one exclosure, and four (males and females) in the other exclosure.

Figure 7. Black-footed tree-rat captured on camera-trap inside one of the fenced exclosures.

Table 2. Number of detections of mammals captured on fence monitoring camera traps between Nov 2013 and Dec 2015. Captures are of animals walking around the outside of the fence unless otherwise stated. No data were available for KAP02 in May 2014.

KAP02													
Year	**Month**	**Northern Brown Bandicoot**	**Black-footed Tree-rat (inside)**	**Northern Quoll**	**Echidna**	**Echidna (inside)**	**Cat (inside)**	**Cat**	**Dingo**	**Pig**	**Water Buffalo**	**Horse**	**Macropus sp.**
2013	Dec	**2**	-	-	1	-	-	-	4	-	-	-	41
2014	Jan	-	-	-	-	-	-	-	1	-	-	-	15
	Feb	-	-	-	-	-	-	-	3	-	-	-	1
	Mar	**2**	-	-	-	-	-	-	3	2	-	-	12
	Apr	-	-	-	-	-	-	-	3	-	-	-	7
	May	na	na	na	na	na	na	na	na	na	na	na	na
	Jun	-	-	-	-	-	-	-	-	-	-	-	4
	Jul	**1**	-	**1**	-	-	-	4	-	-	-	-	24
	Aug	-	-	-	-	-	-	-	2	-	-	-	20
	Sep	-	-	-	-	-	-	-	1	-	-	-	42
	Oct	-	-	-	-	-	-	1	2	-	-	-	15
	Nov	-	-	-	-	-	-	-	3	-	-	-	12
	Dec	-	-	-	-	-	-	2	5	-	-	-	59
2015	Jan	-	-	-	-	-	-	4	23	1	2	3	13
	Feb	-	-	-	-	-	-	2	30	1	4	1	2
	Mar	-	-	-	-	-	-	1	25	-	-	-	8
	Apr	-	-	-	-	-	-	4	24	-	-	-	25
	May	-	-	-	-	-	-	7	10	-	-	-	61
	Jun	-	**4**	-	-	-	-	6	9	1	-	-	59
	Jul	-	-	-	-	-	-	5	6	-	-	-	70
	Aug	-	-	-	-	-	-	8	4	1	1	-	68
	Sep	-	-	-	-	-	-	1	1	-	-	-	36
	Oct	-	-	-	-	-	-	-	-	-	2	1	22
	Nov	-	-	-	-	-	-	2	2	-	-	-	48
	Dec	-	-	-	-	-	-	-	-	-	-	-	7
KAP03													
2013	Nov	-	-	-	-	-	45	38	14	-	-	-	30
	Dec	-	-	-	-	-	-	12	8	-	-	-	54
2014	Jan	-	-	-	-	-	-	-	3	-	-	-	12
	Feb	-	-	-	-	-	-	-	10	3	-	-	11
	Mar	**2**	-	-	-	-	-	2	8	-	-	-	34
	Apr	**4**	-	-	-	-	-	1	-	-	-	-	21
	May	-	-	-	-	2	-	3	-	-	-	-	-
	Jun	-	-	-	-	2	-	3	-	-	-	-	12
	Jul	-	-	-	3	32	-	4	4	2	-	-	42
	Aug	-	-	-	1	-	-	1	28	2	-	-	53
	Sep	-	-	-	-	-	-	-	11	-	-	-	46
	Oct	-	-	-	-	-	-	2	-	-	-	-	24
	Nov	**1**	-	-	-	-	-	-	1	-	-	-	23
	Dec	-	-	-	-	-	-	-	5	-	-	-	46
2015	Jan	-	-	-	-	-	-	3	16	-	-	-	8
	Feb	**7**	-	-	-	-	-	-	-	-	-	-	1
	Mar	-	-	-	-	-	-	1	3	-	-	-	6
	Apr	-	-	-	-	-	-	-	5	-	-	-	10
	May	-	-	-	-	-	-	2	8	-	-	-	23
	Jun	-	-	-	-	-	-	-	1	2	-	-	40
	Jul	-	-	-	-	-	-	11	1	-	-	-	21
	Aug	-	-	-	-	-	-	1	7	-	-	-	50
	Sep	-	-	-	-	-	-	-	-	-	-	-	20
	Oct	-	-	-	-	-	-	-	2	-	-	-	25
	Nov	-	-	-	-	-	-	4	11	-	-	-	50
	Dec	-	-	-	-	-	-	-	1	-	-	-	11

3 VERTEBRATE RESPONSES TO CAT EXCLUSION

If feral cats have a major predatory suppressive effect on small vertebrate populations then we would expect small vertebrate populations to increase inside the exclosures in the absence of cats. If too frequent fire is also having suppressive effects on small vertebrate diversity, and/or interacting with effects of feral cat predation, then we would also expect to see some vertebrate population responses to the exclusion of fire over time. To test these hypotheses, sampling was undertaken at all sites prior to fence construction to obtain baseline data on mammal and reptile relative abundance and species richness, prior to exclusion of feral cats from the fenced sites, and following fence construction.

3.1 Methods - Mammal and Reptile Sampling

3.1.1 Live trapping

The first round of preliminary sampling occurred between March 2013 and June 2013 with a second round of sampling in November 2013. An evaluation of sampling effort, required to maximise mammal captures whilst balancing logistical constraints, was undertaken during the March and June surveys. During these surveys varying numbers of transects per site, and number of traps per transect, were trialled. A standardised trapping methodology was established prior to the November 2013 survey.

Over the period from fence completion in December 2013 to December 2015, each site was sampled three times per year in the late wet (March/April), mid-dry (June/July) and early wet (Oct/Nov) seasons. Within each site six transects, with permanent trap stations, were established 100 m apart. Mammals were captured using a combination of small cage, Elliott and pitfall traps. Thirty-nine mammal traps (26 Elliott traps, 13 cage traps) were placed along each transect spaced 20 m apart (Figure 8). Four pitfall traps were installed 200 m apart along each transect (Figure 9). Each transect of Elliott, cage and pit traps was sampled for 3 nights during each sampling session, with all transects within a site sampled within a period of 7 days. Captured mammals were weighed, pes and head width were measured, and sex and age-class determined prior to release. A microchip (radio frequency identification device) was implanted between the shoulder-blades of medium-sized mammals to allow individual identification. Small-sized mammals were originally marked using non-toxic ink but due to the loss of marks in wet weather, a small fur clip was taken. A mixture of peanut butter, oats and honey was used as bait in cage and Elliott traps. Pitfall traps were checked twice daily and the species and number of individuals captured were recorded. Unequal trapping effort occurred between some sampling periods due to inclement weather and/or problems with meat ants and fire on two occasions. For example only half of the pitfalls were opened at three sites during the first sampling session (November 2013) due to heavy rainfall.

3.1.2 Camera Trapping

Five passive infrared cameras (camera trap) were placed within each site concurrently with live trapping commencing from November 2013 and finishing December 2015. One camera was placed in the centre of each site, and the remaining cameras placed approximately 100 m from this point and equidistantly from each other. Cameras remained deployed for a minimum of 4 weeks.

Each camera was placed at a height of 70 cm above the ground, measuring to the top of the camera housing. A bait station constructed with an 80 mm length of 50 mm PVC pipe and ventilated end caps, to allow scent to escape, was placed approximately 300 mm above the ground at each camera station. A mixture of peanut butter, oats and honey was used as bait. Two of the five site cameras were set 1.5 m from bait stations, and the other three cameras were set 2.5 m from bait stations. Cameras were angled slightly downward so the bait station was in the middle of the image (Gillespie *et al*. 2015).

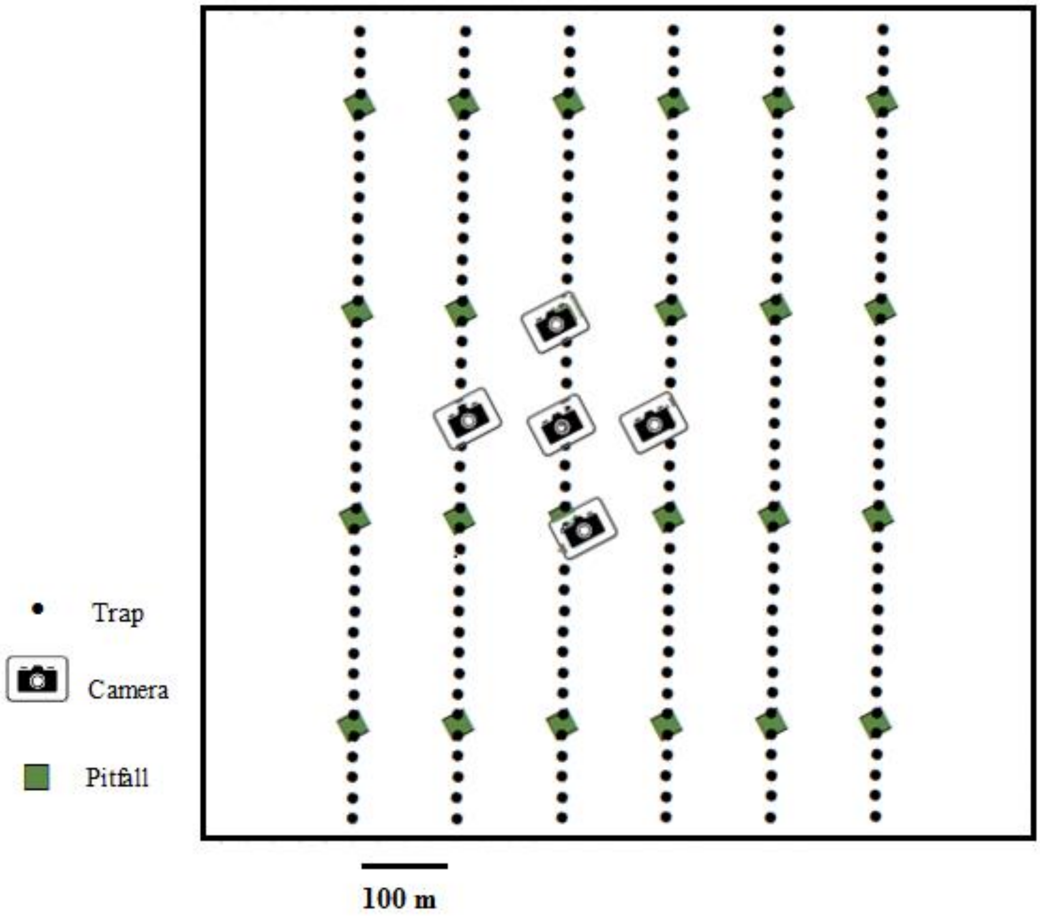


Figure 8. Trapping arrangement at each of the six experimental sites at Kapalga, Kakadu National Park. Black dots designate cage and Elliott traps, green squares designate pitfall traps, and camera icon designate camera traps.

Figure 9. Roy Tipiloura setting up a pitfall trap at Kapalga.

3.2 Statistical Methods

3.2.1 Analysis of Mammal Data

As unequal trapping effort occurred between some sampling periods, capture rates (number of captures per 100 trap-nights) was calculated for each site in each sampling session. For analyses, native mammals were categorised into two groups; small-sized mammals for species with adult body mass < 250 g, and medium-sized mammals for those species with an adult body mass > 250 g. The rationale for this categorisation was based on the observation that the predator fencing excluded medium-sized mammals and therefore they were excluded from analysis of the fence treatment effect. Small-sized mammals were caught using all three trap types, and thus trap effort for all methods was used to derive capture rates. Medium-sized mammals were only captured in wire cage traps and Elliott traps and therefore pitfalls were excluded from capture rate calculations for this group.

Data for capture rates were non-normally distributed, and zero-inflated (78%), with large variation in error between sampling sessions. For these reasons, a non-parametric ANOVA with repeated measures (Friedman test) was used to analyse the data. Since fire affected all sites at some point during the project, the treatment variable was reduced to two classes: fenced verses unfenced. To compare differences among repeated measures of small-sized mammal capture rates between fenced and unfenced treatments, mean capture rate was calculated for fenced and unfenced sites in each sampling session from March 2013 to December 2015.

3.2.2 Analysis of Reptile Data

Reptile captures recorded from pitfalls were used in analyses. Where taxonomic identification was uncertain, taxa were collated to genus level. The number of individuals and number of species of reptile captured at each pitfall within each sampling session between November 2013 and November 2015 were calculated and used as response variables to model treatment effects. Data collected prior to the November 2013 sampling session were not included in the analyses due to inadequate information recorded regarding sampling effort and recent fire history at each pitfall.

As the study design incorporated repeated measures at pitfalls nested within sites, and counts, which typically follow a Poisson distribution, were used as response variables, we analysed data using generalized linear mixed models (GLMMs; Pinheiro and Bates 2000). GLMMs provide a means for the inclusion of random effects; therefore the variance induced by site-effects could be estimated as a source of random variation in the data. To assess the effect of predator exclusion on mean reptile abundance and mean species richness over time we analysed the data with the R package ‘lme4’ (Bates *et al.* 2015) using the ‘bobyqa’ optimizer to minimise issues with model convergence.

In order to account for the variation in sampling effort between pitfalls in each sampling session, the number of 12 hour sampling intervals undertaken at each pitfall in each sampling session was $_n$log transformed and used as an offset in the models to account for this variation (Zuur *et al.* 2009). Therefore, interpretation of the modelled response is the expected number per unit effort (12 hour sampling interval). Since fire affected all sites at some point during the project, the treatment variable was reduced to two classes: fenced verses unfenced (Table 3). To test for the effect of fire, the number of months since a fire (TSF) occurred within 10 m from the pitfall was recorded for each pitfall in each sampling session and included in models. TSF was transformed by adding a constant and then taking the square root to improve normality. Seven sampling sessions were undertaken between November 2013 and December 2015; a time variable was created to account for the change in response over time.

The modelling approach involved building a model including the fixed effects of Time, Fence treatment and their interaction, and the additive effect of TSF. A series of models was run which included and excluded random effects, so that the validity of including random effects could be assessed. Random effects models included intercepts which were allowed to vary for each site and for each pitfall within each site. Furthermore, because the relationship between abundance/species richness and time is likely to be different for each site, due to inherent site-effects, the slope for each site was allowed to vary. Model fit was compared using Akaike Information Criterion (AIC) and likelihood ratio test to select the optimal model. Model residuals were plotted against the fitted values and each explanatory variable to identify violation of homogeneity. A smoother was applied to the TSF variable using the R package ‘mgcv’ (Wood 2004) and plotted against model residuals to check for non-linear patterns. The fixed effect of sampling season was added to the model fitting reptile abundance as structured heterogeneity was identified in the residual plots. An additional model was run to assess the effect of the season of sampling on species richness. Plots of the variable effects were calculated and plotted using R package ‘effects’ (Fox 2003).

Little is known of the ecology of the endangered Yellow-snouted Gecko *Lucasium occultum* (endangered, *EPBC* Act) which is restricted to the Mary and West Alligator River catchments. Although dynamic models, which can evaluate underlying processes in changes of occupancy over time, are most useful in understanding local colonisation and extinction patterns; due to the paucity of capture data this could not be undertaken for the Yellow-snouted Gecko. Furthermore there was inadequate data to undertake single-season occupancy models on each sampling session. Whilst occupancy models are sensitive to the assumption of closure, if changes in occupancy over the survey time period are random, then occupancy estimates correspond to the probability of an area being used, and detection probability corresponds to the product of the probability of presence and probability of detection given presence (MacKenzie *et al.* 2006).

To obtain a general estimate of site occupancy over the study period, and the potential effects of TSF, captures at pitfalls within a sampling session were collapsed to presence-absence data for each session. Static site-occupancy models were run using the R package ‘unmarked’ (Fiske and Chandler 2011), incorporating Site and TSF as covariates for occupancy and detection, respectively. The following four models were fitted to the data: (1) all parameters were held constant [ψ(.)p(.)]; (2) occupancy held constant and detection probability may vary by TSF [ψ(.)p(TSF)]; (3) occupancy differs for each site and detection probability held constant [ψ(.)p(TSF)]; and (4) occupancy differs for each site and detection probability may vary by TSF [ψ(Site)p(TSF)]. AIC were used for model selection (Burnham and Anderson 2002), and a parametric bootstrap method was used to check the adequacy of the fit of the most parameterised model to the data.

Table 3. Fixed and random variables used in candidate set of models investigating the effect of predator exclusion and time since fire on reptile species richness and number of captures.

Variable	**Type**
Fixed Effects	
Treatment	Categorical (fenced, unfenced)
Time (Sampling Sessions)	Continuous (range: 1 – 7)
Time Since Fire (TSF)	Continuous (range: 0 – 54 months)
Season	Categorical (Dry, Early Wet, Late Wet)
Random Effects	
Site (intercept and slope)	6 Sites
Pitfall (intercept)	144 pitfalls (24 pitfalls/ site)

3.3 Results

3.3.1 Mammals

Between March 2013 and November 2015, 33,774 trap-nights were deployed using small cage and Elliott traps, and 3,476 trap-nights using pitfalls over eight survey sessions. Total capture rates (captures per 100 trap-night) of small- and medium-sized mammals across the six sites were extremely low throughout the experiment, with capture rates across groups ranging from 0 to 4 (small-sized mammals: mean = 0.16, S.E. = 0.9; medium-sized mammals = 0.17, S.E. = 0.05) across all sites and surveys (Figure 11). The highest capture rates were encountered in November 2013, which coincided with heavy rainfall.

Six species of native mammal were trapped across the six sites (Table 4); Northern Quoll, Northern Brown Bandicoot, Common Brushtail Possum, Grassland Melomys *Melomys burtoni*, Dusky Rat *Rattus colletti*, and Delicate Mouse *Pseudomys delicatulus*. One sub-adult *Pseudomys* species was captured which could not be confidently identified to species level. No more than five species were trapped on any one site; three species were trapped on four of the six sites.

There were a total of 115 mammal captures over the eight sessions, including 56 small-sized mammals and 59 medium sized-mammals. Seventeen individual Northern Quoll, including 3 lactating females, 12 Northern Brown Bandicoot and two Brushtail Possum were captured. Low numbers of quolls and bandicoots were trapped within the fenced sites during the baseline surveys but were never trapped within the fenced sites following fence completion. Small mammals were trapped at all study sites during at least one survey between November 2013 and November 2015. Dusky Rat and Grassland Melomys were only captured during early or late wet season surveys, consistent with population movements of these species into savanna woodland during lowland seasonal inundation (Redhead 1979). There were no significant differences detected on repeated measures between fenced and unfenced sites with respect to small mammal capture rates ($\chi^2 = 0.67$, $df = 1$, $p = 0.41$).

No small mammals were detected on camera-trap on any of the sites. Bandicoots were most often detected on sites using camera traps (Table 5), whilst on at least half of the occasions quolls and possums were detected concurrently with live-trapping and camera-trapping. In May 2015, a Black-footed Tree-rat was detected on a fence-monitoring camera trap (Table 2), but this species was never detected during targeted native mammal sampling. Quolls and bandicoots were detected outside the fenced sites on several occasions. Dingos and cats were often detected on the same site in the same sampling session as medium-sized mammals. Cats were detected across non-fenced sites in all but one sampling session, and dingos on five of seven sampling sessions (Figure 10).

Table 4. Number of native mammals live-trapped represented as number of captures and number of identified different individuals.

Common Name	Species	Captures	Individuals
Northern Quoll*	*Dasyurus hallucatus*	34	17
Northern Brown Bandicoot	*Isoodon macrourus*	23	12
Grasslands Melomys	*Melomys burtoni*	26	
Delicate Mouse	*Pseudomys delicatulus*	3	
(Delicate Mouse or Kakadu Pebble-mound Mouse)	*Pseudomys* sp.	1	
Dusky Rat	*Rattus colletti*	26	
Brushtail Possum	*Trichosurus vulpecula*	2	2

* Endangered, *EPBC* Act

Table 5. Proportion of detections of species, across all sites and sampling sessions, where a species was detected only using either camera-trap or live-trapping or by both methods concurrently.

Survey Method	Northern Brown Bandicoot	Northern Quoll	Brushtail Possum
Camera-trap	58%	31%	0%
Live-trapping	0%	13%	50%
Both Camera-trap and Live-trapping	42%	56%	50%

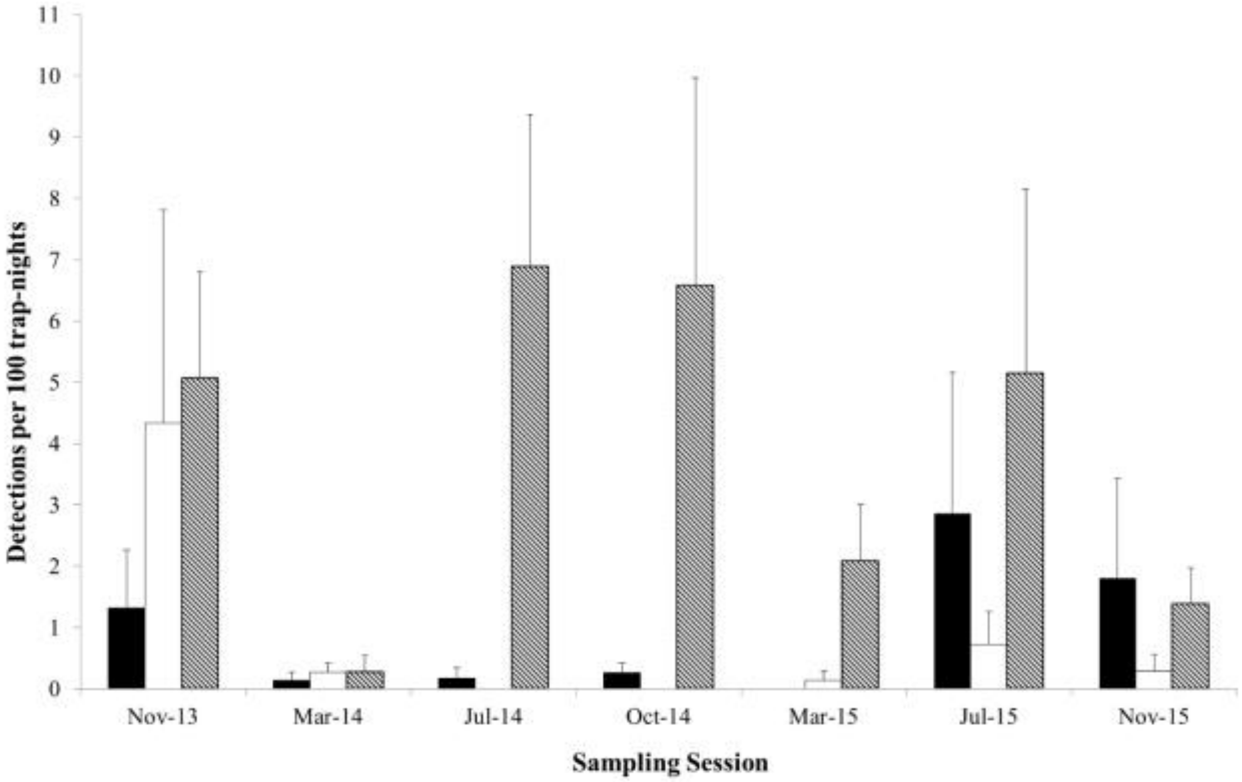
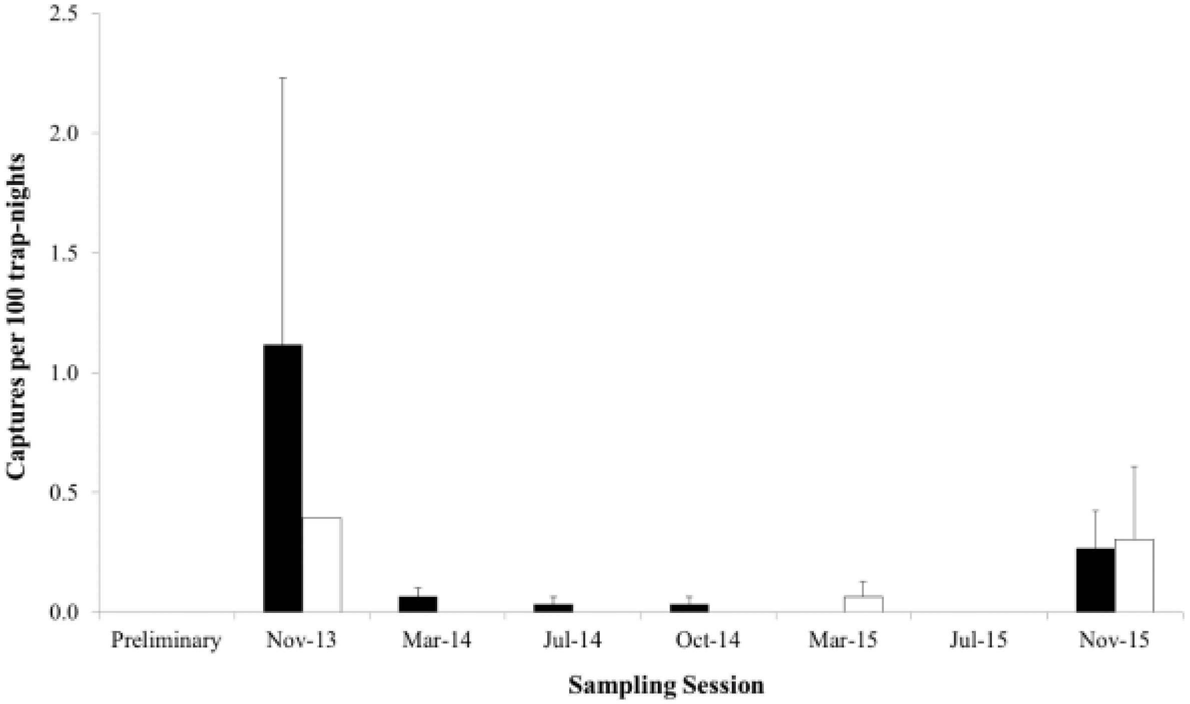


Figure 10. Mean number of detections per 100 trap-nights of cats (solid bars), dingos (open bars) and medium-sized mammals (hatched bars) captured on camera trap on repeated sampling across non-fenced sites (n = 4). Error bars represent standard error.



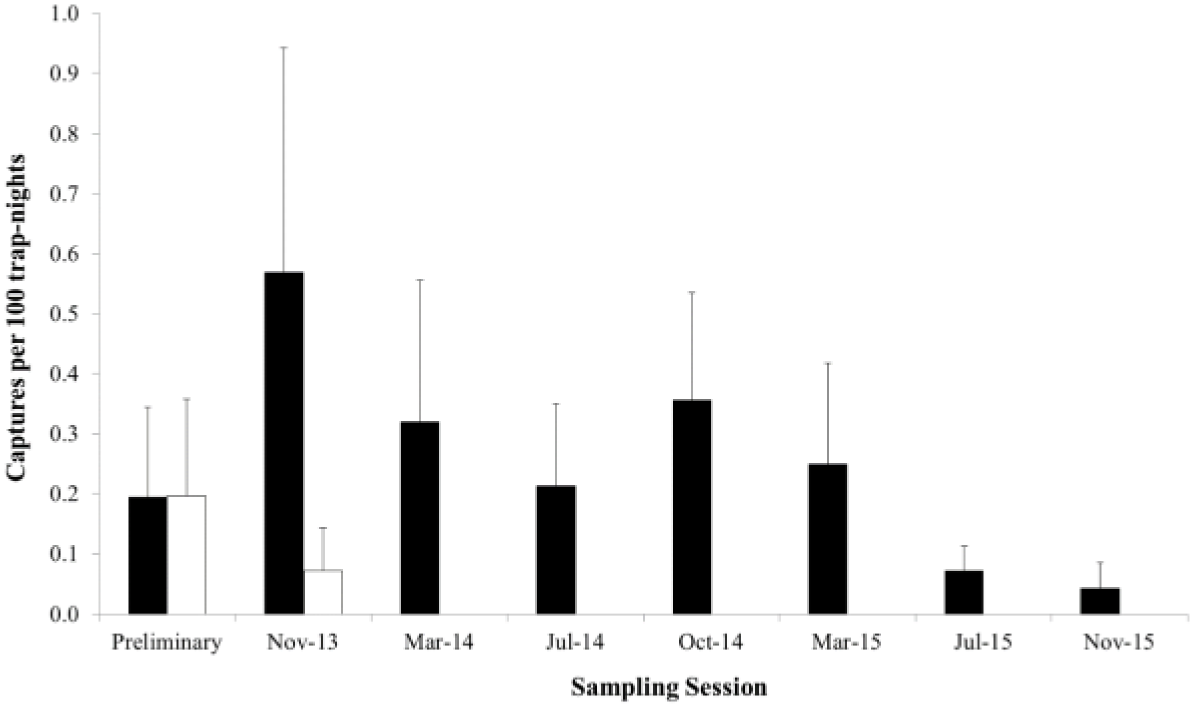


Figure 11. Mean trap rate (captures per 100 trap-nights) of small-sized native mammals (top) and medium-sized mammals (bottom) between the two treatments by sampling interval. Solid bars are unfenced sites (n = 4), open bars are fenced sites (n = 2). Error bars represent standard errors.

3.3.2 Reptiles

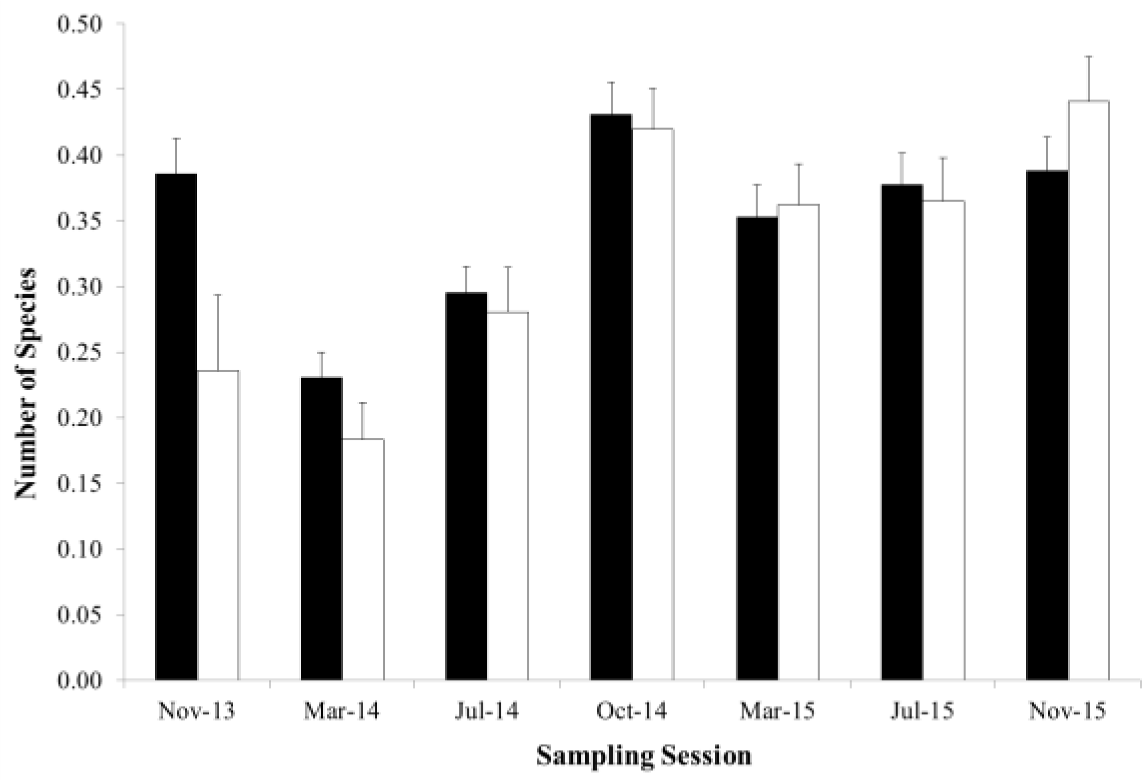
Thirty-one species of reptile were captured in pitfalls across the sites, and one species was only ever captured in Elliott traps (Table 6). Mean number of reptiles captured and the mean number of species detected varied between sampling sessions and between treatments within sampling sessions with the highest means obtained in early wet sessions (Figure 12). Ten species of amphibian across eight genera were also captured (Table 6).

Table 6. Reptile and amphibian species captured at the Kapalga study sites between March 2013 and December 2015.

Skinks		**Snakes**	
	Carlia amax		*Furina ornata*
	Carlia gracilis		*Cryptophis pallidiceps*
	Carlia munda		*Ramphotyphlops cf. tovelli*
	Carlia triacantha		*Ramphotyphlops unguirostris*
	Cryptoblepharus sp		*Vermicella intermedia*
	Ctenotus borealis	**Geckos**	
	Ctenotus essingtonii		*Amalosia rhombifer*
	Ctenotus kurnbudj		*Gehyra australis*
	Ctenotus storri		*Heteronotia binoei*
	Eremiascincus isolepis		*Lucasium occultum**
	Glaphyromorphus darwiniensis		*Lucasium stenodactylum*
	Lerista karlschmidti		*Oedura marmorata^*
	Menetia greyii	**Frogs**	
	Morethia storri		*Crinia sp*
	Notoscincus ornatus		*Cyclorana longipes*
	Proablepharus tenuis		*Cyclorana australis^*
Legless-Lizards			*Limnodynastes convexiusculus*
	Delma borea		*Litoria bicolor*
	Lialis burtonis		*Litoria caerulea^*
Monitors			*Litoria nasuta*
	Varanus scalaris		*Litoria tornieri*
	Varanus tristis		*Notaden melanoscaphus*
Dragons			*Platyplectrum ornatum*
	Diporiphora bilineata		*Rhinella marina*
			Uperoleia sp

*Endangered, *EPBC* Act

^Species only captured in Elliott or cage traps



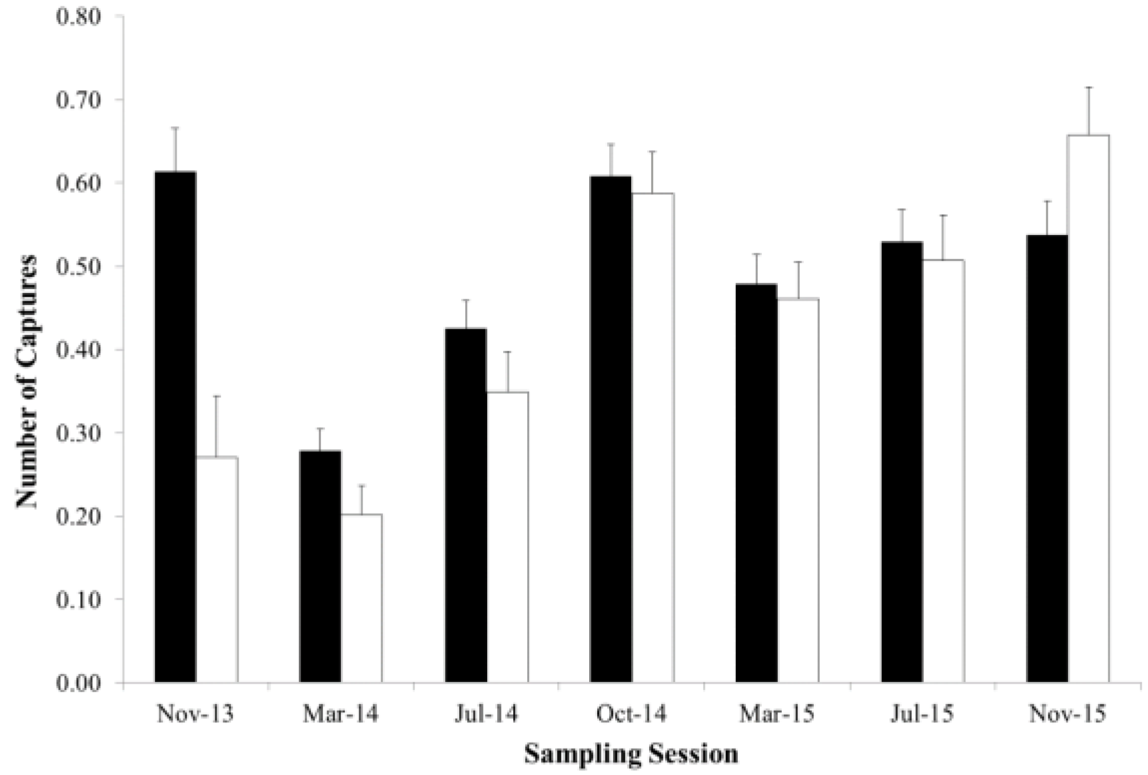


Figure 12. Mean number of reptile species (top) and mean number of captures of reptiles (bottom) per pit by treatment during surveys undertaken at Kapalga study sites. Values have been standardised by the number of 12 hour sampling intervals pitfalls were open. Solid bars are control sites (n = 4), open bars are fenced sites (n = 2). Error bars represent standard error.

A model incorporating a random intercept for pitfall-effects and a random intercept and random slope for site-effects was the best supported model structure for describing variability in reptile abundance. The effect of season was added to the model to improve homogeneity of residuals and significantly reduced deviance ($\chi^2 = 66.7$, *d.f.* = 2, $p < 0.001$). Fencing to exclude predators had a significant positive effect on reptile abundance over time ($p = 0.044$) and there was a significant additive effect of time since fire but the effect was small ($p = 0.039$) (Table 7). Reptile abundance within fenced sites was lower than at unfenced sites at the commencement of the experiment, but there was a greater increase in abundance over time at fenced sites (Figure 13). The effect of excluding predators resulted in a doubling of mean reptile abundance over two years, while mean reptile abundance remained constant at unfenced sites (Figure 14).

There was greater variation in abundance amongst sites within treatments than amongst pitfalls within sites (site variance about the intercept = 0.077, pitfall variance about the intercept = 0.040). There was minimal correlation between repeated observations at pitfalls (*Intra-class correlation* = 0.001), but model deviance was significantly reduced with pitfall-effect included.

Table 7. Model results for the fixed effects from a generalised linear mixed model testing the response of mean reptile abundance to predator exclusion over time. The intercept term represents reptile abundance response for unfenced sites in the dry season. The expected abundance is per unit effort (12 hr sampling interval). Bold values represent significant terms.

Model:

$$Abundance = exp(B_0 + B_1(Time) + B_2(Treatment) + B_3(Season) + B_4(sqrtTSF) + B_5(Time \times Treatment) + offset(_n logEffort))$$

	Estimate	*S.E.*	*p-value*
(Intercept)	-0.965	0.167	**< 0.001**
Time	0.003	0.032	0.935
Fence	-0.567	0.272	**0.037**
Season: Early Wet	0.229	0.046	**< 0.001**
Season: Late Wet	-0.191	0.055	**0.001**
sqrt(TSF)	0.040	0.019	**0.039**
Time x Fence	0.112	0.056	**0.044**

A model incorporating a random intercept and random slope for site-effects was the best supported model structure for modelling reptile species richness. Variation in species richness amongst pitfalls within sites was close to zero and there was no correlation between repeated observations at pitfalls. Species richness within fenced sites in this experiment was lower than at unfenced sites at the commencement of the experiment but fenced sites had a greater rate of increase in species richness over time (Figure 16), but the observed effect was not significant when seasonal variation was taken into account.

The effect of time since fire was not a significant variable in the model explaining species richness ($p = 0.56$) and therefore a model was re-run without this term. Exclusion fencing had a significant positive effect on reptile species richness over time (*p* = 0.031) (Table 8A). There was variation in richness amongst sites within treatments (site variance about the intercept = 0.01) and variation in the response in species richness change over time amongst sites (site variance around the slope = 0.001). When the effect of season was added to the modelled response of species richness, the interaction of time and fence treatment became non-significant (*p* = 0.062) (Table 8B). The model including season significantly reduced model deviance (χ^2 = 26.0, d.f. = 2, *p* < 0.001) and therefore had greater support.

Table 8. Model results for the fixed effects from a generalised linear mixed model testing the response of mean reptile species richness to predator exclusion fencing over time. The intercept term represents species richness response for unfenced sites (A) and unfenced sites in the dry season (B). Bold values represent significant terms. Expected richness is per unit effort (12 hr sampling interval).

A.

Model:

Species Richness = exp(B0 + B1(Time) + B3(Treatment) + B4(Time x Treatment) + offset(nlogEffort))

	Estimate	*S.E.*	*p - value*
(Intercept)	-1.175	0.084	**< 0.001**
Time	0.031	0.023	0.183
Fence	-0.467	0.163	**0.004**
Time x Fence	0.091	0.042	**0.031**

B.

Model:

Species Richness = exp(B0 + B1(Time) + B2(Treatment) + B3(Season) + B4(Time x Treatment) + offset(nlogEffort))

	Estimate	*S.E.*	*p-value*
(Intercept)	-1.194	0.085	**< 0.001**
Time	0.022	0.022	0.319
Fence	-0.381	0.150	**0.011**
Season: Early Wet	0.188	0.054	**0.001**
Season: Late Wet	-0.096	0.064	0.132
Time x Fence	0.075	0.040	0.062

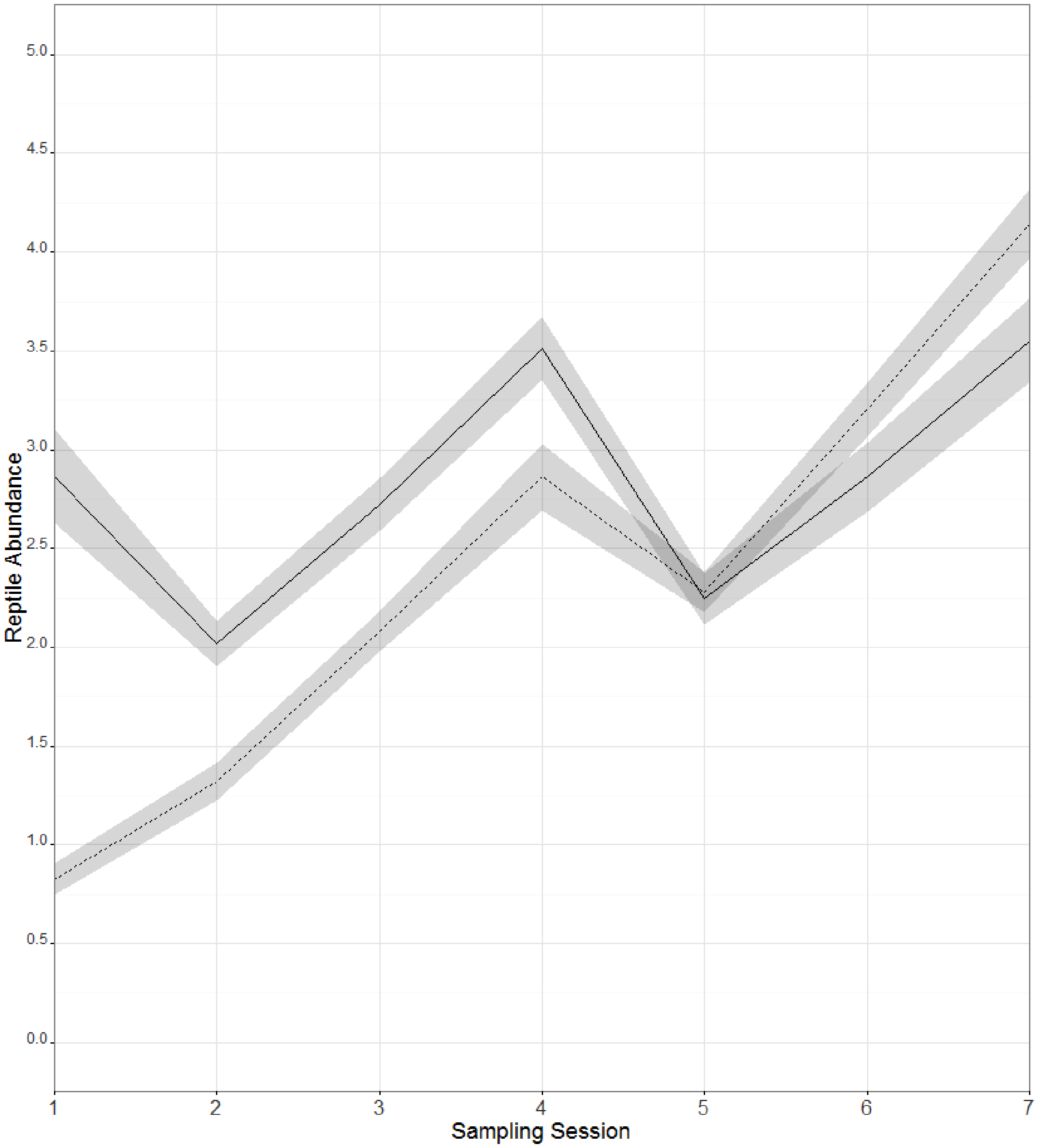


Figure 13. Plot of the mean observed linear trend in reptile abundance for both unfenced (solid line) and fenced (dotted line) sites. The values are model fitted means to the observed Kapalga site data. Shaded area represents 95% confidence interval about the mean. Note that session 1 immediately precedes fence completion.

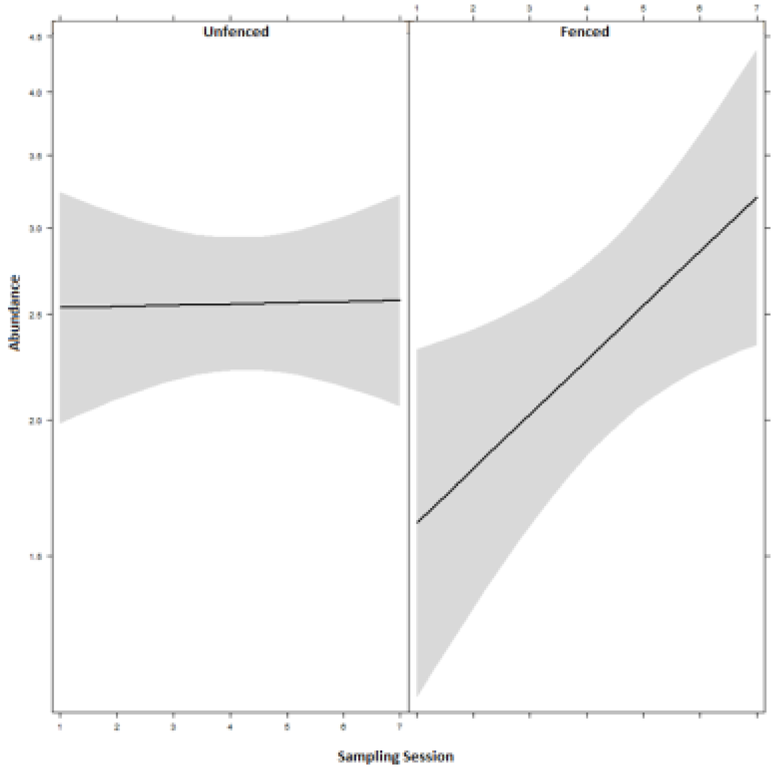


Figure 14. Plot of treatment effect on reptile abundance over time, unfenced (left) and fenced (right). Solid black line represents the estimated effect and the shading represents 95% confidence intervals.

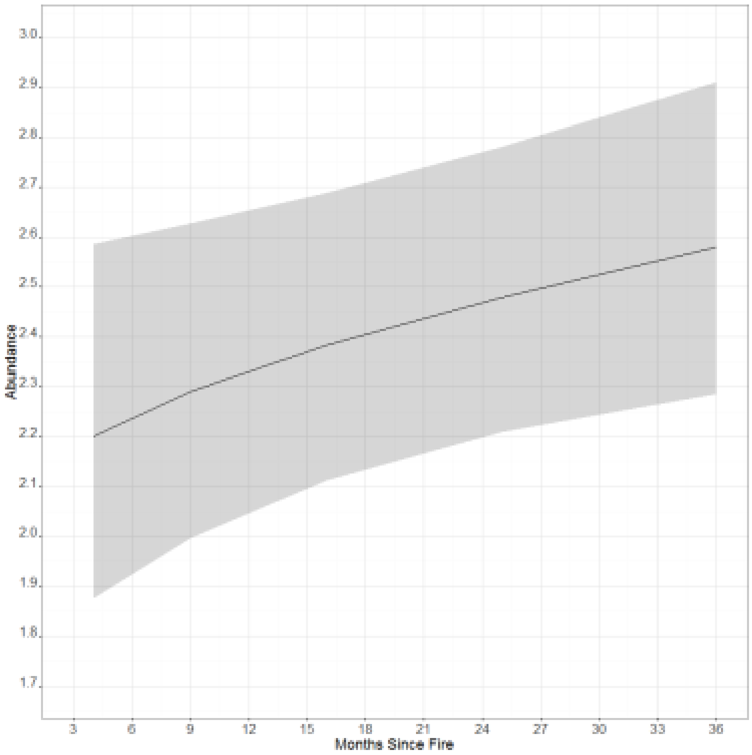


Figure 15. Plot of fire effect on reptile abundance. Solid black line represents the estimated effect and the shading represents 95% confidence intervals.

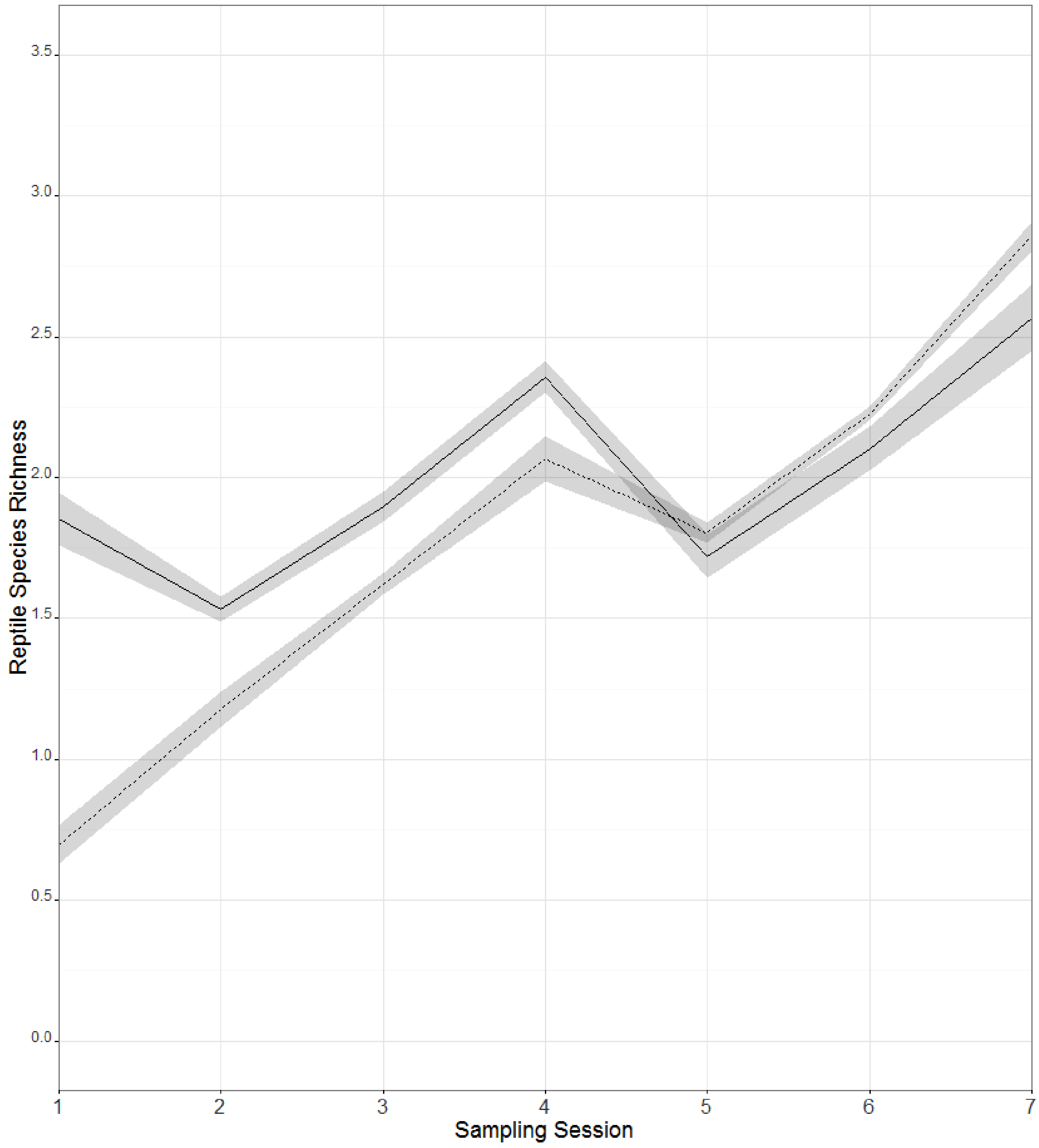


Figure 16. Plot of the observed linear trend in reptile species richness for both unfenced (solid line) and fenced (dotted line) sites. The values are model fitted means to the observed Kapalga site data. Shaded area represents 95% confidence interval about the mean. Note that session 1 immediately precedes fence completion.

The seasonal effect was significant in models for both species richness and reptile abundance. Reptile abundance and species richness were greatest in the early wet season (October/November) and lowest in the late wet season (March/April) (Figure 17).

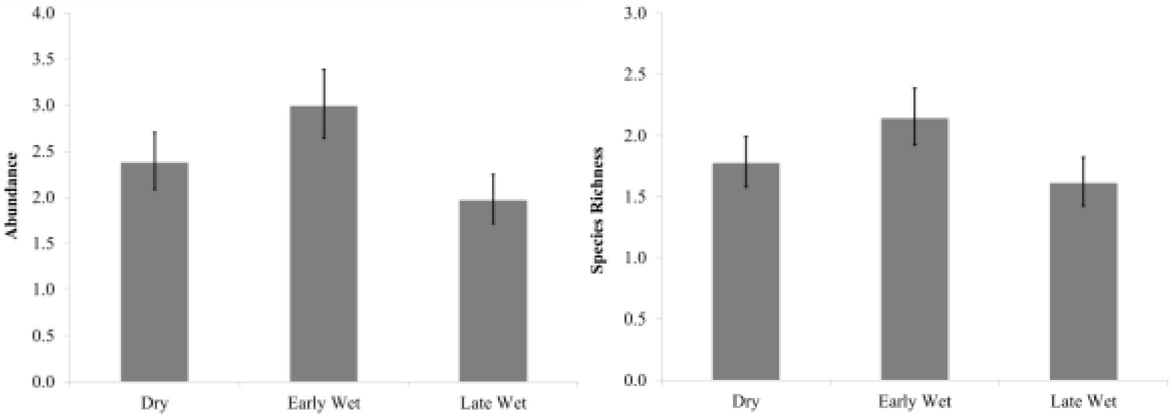


Figure 17. Seasonal effect on mean reptile abundance (left) and mean species richness (right). Error bars are 95% confidence intervals.

Treatment effect on taxonomic groups of reptiles could not be analysed utilising the same modelling approach due to zero-inflated data (> 25 % of the data). Alternate modelling approaches will be explored in the future to assess taxonomic responses to fire and predator exclusion.

A total of 36 captures of the threatened Yellow-snouted Gecko *Lucasium occultum* were recorded at 22 pitfalls across four sites. An occupancy model evaluating study site occupancy and effects of time since fire on the detection process was well supported with 92% of the AIC weighting (Table 9). Most captures occurred on two sites (86% of captures) and consequently the probability of any pitfall on these sites being occupied during the study was greater than 0.65 (Table 10). The probability of detecting a Yellow-snouted Gecko at a pitfall on repeated sampling sessions was lowest immediately following fire (detection probability = 0.05, SE = 0.02, p = 0.01) and increased with time since fire (Figure 18), suggesting that fire could negatively influence this species.

Table 9. Model selection results of site occupancy by Yellow-snouted Gecko.

Model		AIC	ΔAIC	AICwt
ψ(Site)p(TSF)	8	232.60	0	0.92
ψ(Site)p(.)	7	237.62	5.02	0.08
ψ(.)p(TSF)	3	263.36	30.76	0.00
ψ(.)p(.)	2	264.68	32.08	0.00

Several larger reptiles were observed in the Kapalga study area including: Frilled-necked Lizard *Chlamydosaurus kingii*, Olive Python *Liasis olivaceus*, Merten's Water Monitor *Varanus mertensi*, Floodplain Monitor *Varanus panoptes*, Slaty-grey Snake *Stegonotus cucullatus*, and Keelback *Tropidonophis mairii*. However, data on these larger species were too sparse for statistical evaluation.

The introduced Cane Toad *Rhinella marina* was detected at all sites during at least one sampling session with most detections occurring at the end of the wet season. Cane toads were captured in small cage traps (25% of captures), Elliott traps (13%) and pitfall traps (62 %). There was no difference in the mean number of captures of cane toads between treatments (Friedman test: $\chi^2 = 0$, $df = 1$, $p = 1$) (Figure 19), and the fences were not a barrier to their movement.

Table 10. Capture rate (number of captures/effort x 100) of Yellow-snouted Gecko trapped in pitfall traps at each site in each sampling session undertaken at Kapalga study sites. Last row of values represent total number of captures. Last column represents probability of occupancy, and standard error in parenthesis, at pitfalls within each site. (KAP02 and KAP03 were fenced sites).

Site	Nov-13	Mar-14	Jul-14	Oct-14	Mar-15	Jul-15	Nov-15	Site Captures	Occupancy (SE)
KAP01	0	0	0	0	0	0	0	0	0 (0)
KAP02	0	0	2.2	2.2	2.1	2.8	1.4	15	0.66 (0.22)
KAP03	0	0	0	1.5	0	0	1.5	4	0.15 (0.11)
KAP04	0	0	0	0	0	0	0.7	1	0.07 (0.07)
KAP05	0	0	0	0	0	0	0	0	0 (0)
KAP06	2.4	0	0.7	0.7	3	0	4.9	16	0.72 (0.22)
Captures per Sampling session	3	0	4	6	7	4	12	36	

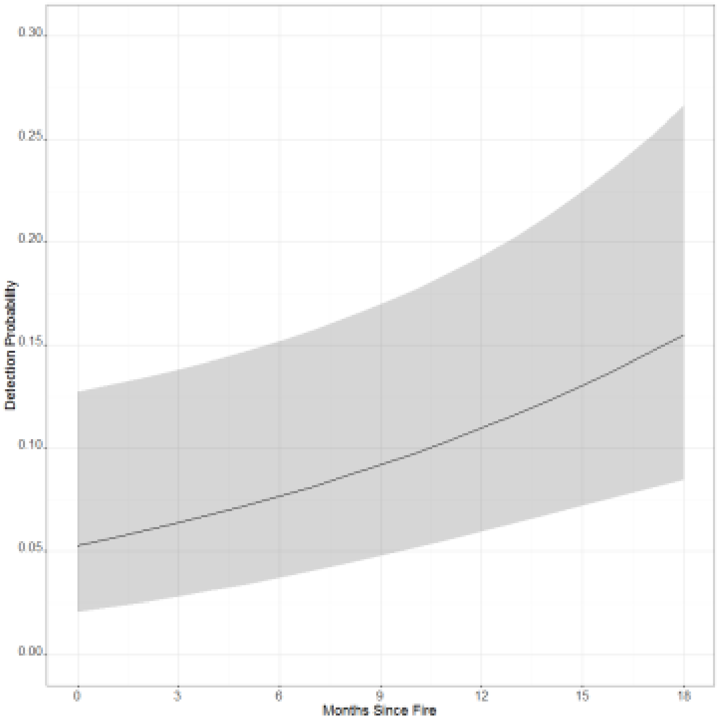


Figure 18. Effect of Time Since Fire on the detection probability of Yellow-snouted Gecko trapped in pitfall traps within a 3-day sampling session. The solid line represents the predicted value. Shaded area represents 95% confidence interval.

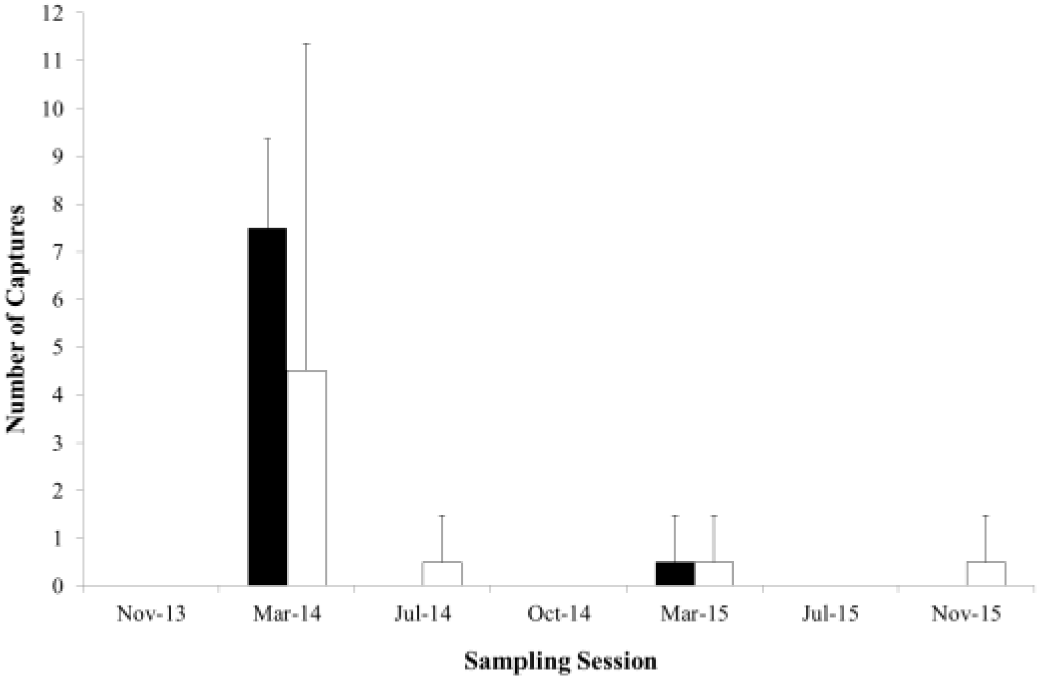


Figure 19. Mean number of captures of cane toads trapped by treatment during surveys undertaken at Kapalga study sites. Values represent captures from pitfalls, Elliott and cage traps. Solid bars are control sites (n = 4), open bars are fenced sites (n = 2). Error bars represent upper 95% confidence intervals.

4 PREDATOR ECOLOGY

4.1 Methods

4.1.1 Cat Tracking

After fence construction was completed on the first exclosure, a cat was observed on camera trap within the exclosure. It was presumed that this animal became trapped inside the fenced area when fence construction was completed. Eight large cage traps (30 x 30 x 90 cm) were set along the inside of the fence and in front of the camera traps used to monitor predator incursion (see Section 2.6). Cage traps were left open with tinned cat food (sardines and pilchards) provided for several days until camera traps confirmed the cat was entering the cage traps. Within 10 days of fence completion, the male cat was trapped with a cage trap. A GPS collar (Telemetry Solutions Quantum 4000) was attached (Figure 20), and the cat was released outside the exclosure. The GPS unit was programmed to record location fixes every 15 minutes for two-day bouts, starting and finishing at 12 pm, followed by a five-day interval of four fixes per day (2 am, 6 am, 6 pm, and 10 pm). The GPS collar remained on the cat for 57 days between November 2013 and Jan 2014. The area used by the cat was estimated using a convex hull with XTools Pro 11.1 extension in ArcGIS v 10.2.

In June 2015, a trial was undertaken to live-trap more feral cats in order to obtain home range data on a larger sample of cats. Twenty cage traps and 20 leg-hold traps (Victor soft-jaw size #1.5 and #1.75) were placed in areas frequented by cats such as road verges, intersections and clearings. Cage traps were scent-baited with catnip spray and rat/mouse bedding. Leg-hold traps were deployed in pairs and covered with a light dusting of dirt. A bower was built around each pair of leg-hold traps to minimise captures of non-target species (Figure 21). Leg-hold traps were scent-baited with cat urine. Telemetry trap-checking devices were attached to the leg-hold traps to regularly monitor captures remotely. Cage and leg-hold traps were visually checked twice a day.

Figure 20. GPS collared cat at Kapalga. (Photo Stuart Young)

Figure 21. Leg-hold trap set up within a bower at Kapalga. (Photo Danielle Stokeld)

4.1.2 *Density Estimates of Cats*

To estimate the density of cats in the Kapalga area two grids of 48 camera traps each were deployed in June 2015 (Figure 22). Each grid consisted of three transects spaced 500 m apart and approximately 7 km long. Sixteen cameras were deployed along each transect and spaced 400 - 600 m apart, except where landscape features prohibited (e.g. billabongs, fenced areas). Cameras were set at a height of 45 cm above the ground (measured to the top of the camera housing) and angled slightly down from horizontal. Cat litter soiled with cat urine was used at every second camera position as a potential attractant. Where possible, cameras were placed either along roads/tracks, at ecotones such as floodplain and billabong edges, or natural clearings. White light flash cameras (Reconyx PC850 and HC550), which take colour photos by day and night, were used at all camera locations to improve individual cat identification. Camera traps were deployed for five weeks.

Three people independently assessed camera trap images and assigned each discrete detection of a cat to a unique individual based on coat colour and pattern variation. Images of poor quality that could not be confidently assigned to an individual cat were not included in analyses. Results from each observer were collated and each discrete detection was assigned to an individual cat when there was consensus across all three observers. A daily capture history was generated for each individual for analysis.

We estimated cat density using spatially-explicit mark-recapture (SECR) analysis implemented in the R package ‘secr’ (Efford 2015). In SECR modelling, a hierarchical model is implemented to estimate animal density. The probability of capture (detection) is modelled as a function of the distance between an individual’s activity centre and the camera trap location where an individual was photographed, and density is estimated with a Poisson point process model (Efford *et al.* 2004). As inaccessible areas in the landscape affect the geometry of the home range of animals, a polygon was developed in ArcGIS which defined the boundary of accessible habitat around the camera trapping grid and was used to restrict the state-space in the model. This approach was employed because the South Alligator River and the West Alligator River were considered to be hard boundaries for cats.

The capture probability, as a function of distance, can take a myriad of functional forms (Royle *et al.* 2014). To assess which functional form best fitted the observed data, a set of models was run using different functional forms (half-normal, exponential, hazard-rate). The functional form which had the smallest Akaike Information Criterion (AIC) value was used in subsequent models. The sensitivity of density estimates to changes in the buffer width, the maximum distances from each individual’s home-range centre where detection was zero, were assessed by fitting a series of models with varying buffer widths and assessing stability of the likelihood and influence on density estimates. Density estimates stabilised at a buffer width of 7000 m, which was then used in all subsequent models.

Capture probability can be influenced by various factors, including temporal and behavioural responses, and random variation among individuals (individual heterogeneity). Therefore, we developed a set of models with different covariates influencing the detection parameter of cats (*g0,* the probability of detection at the home range centre; σ, the spatial scale parameter), in order to assess the following:

- influence of using urine as an attractant.
- influence of placing camera-traps along vehicle tracks.

- learned response (model ‘b’) where the probability of detection of an individual at all sites changes dependent on previous capture history.
- learned response by site (model ‘bk’) where probability of detection of an individual changes at a particular site once it has been captured on camera.

Candidate models were ranked using AIC corrected for small sample sizes (AICc) and calculating AIC weights (Burnham and Anderson 2002). Models with ΔAICc values ≤ 2 from the most parsimonious model were strongly supported.

4.1.3 *Predator Diet Analysis*

Surveys to collect predator scats were undertaken concurrently with five trapping sessions, from July 2014 to November 2015. Forty-two kilometres of vehicle track in the Kapalga study area were walked by project personnel visually scanning for dingo/dog, *Canis* cf. *lupus* and cat scats. Each scat located was collected in a paper bag and the GPS coordinates and date recorded. Scats were sun-dried and sent to an authority on scat analysis (B. Triggs, Mallacoota, Victoria) for identification of prey. Results were reported at the genus level where there was ambiguity over species-level identification, or classified and clustered by taxonomic group.

We categorised native mammal species as small, medium, or macropod based on body weight. Species with adult body mass < 250 g were classified as small mammals, while those weighing between 250 g to 5 kg were classified as medium-sized mammals. Animal weights were based on the mean adult body weight published in Van Dyck *et al.* (2013). Frequency of occurrence, defined as the proportion of scats with a particular prey present, was used to assess the prominence of particular prey in the diet of cats and dingos in the Kapalga region.

To compare similarity between the diets of dingoes and cats we calculated dietary overlap of prey species using Pianka’s index (Pianka 1973):

$$\mathrm{DO} = \frac{\sum \mathrm{P}_{ij}\mathrm{P}_{ik}}{\sqrt{\sum \mathrm{P}_{ij}{}^{2} \sum \mathrm{P}_{ik}{}^{2}}},$$

where P_{ij} is the proportion of prey species *i* in the diet of predator *j* and P_{ik} is the proportion of prey species *i* in the diet of predator *k*. Dietary overlap values range from 0 to 1, with 0 indicating no overlap and 1 indicating complete overlap.

4.2 Results

4.2.1 *Cat Observations and Tracking*

Cats were detected on camera trap at five of the six study sites, including a fenced site prior to fence completion. Cats were also observed on camera trap deployments undertaken prior to project commencement in the Kapalga area (Figure 22). The GPS-collared male cat removed from the fenced exclosure used an area of approximately 869 ha (Figure 22). Analysis of home range and fine-scale habitat selection has not been undertaken. No cats, or other species, were captured during the 7-night cat trapping trial.

4.2.2 Density Estimates of Cats

Consensus across three observers of individual cat identification was reached on 99 of 113 discrete cat detections. In total, 22 individual cats were confidently identified from camera trap images across the two trapping grids. No cats were detected using both grids. The best supported model was model ‘bk’ with 97 % of the AICc weighting (Table 11), where the camera-trap-capture probability was associated with learned behaviour at each trap site. Density estimate (± s.e.) based on the best-fitting model was 0.19 ± 0.05 individuals per km^2. Capture probability (if trap to home range centre distance is zero) for an individual at a particular site increased from 0.019 to 0.059 after the first capture. The increase in capture probability on consecutive captures may have been driven by un-modelled variation in camera placement and/or location that influenced cat movements. Models including urine as an attractant, placement on vehicle tracks and the global behavioural model were not competitive and thus did not explain the increase in re-capture. The mean distance cats moved between consecutive capture locations, pooled over individuals, was 981 m (range: 387 – 6402 m).

Table 11. Model comparison using Akaike’s Information Criterion adjusted for small sample sizes (AICc) of spatially-explicit capture-recapture (SECR) models. All models within ten units of the best fit model are shown.

Model	AICc	Delta AICc	Weight
g0(bk),σ(.)	1232.50	0.00	0.95
g0(track), σ(.)	1238.93	6.44	0.04
g0(track), σ(track)	1240.84	8.34	0.01

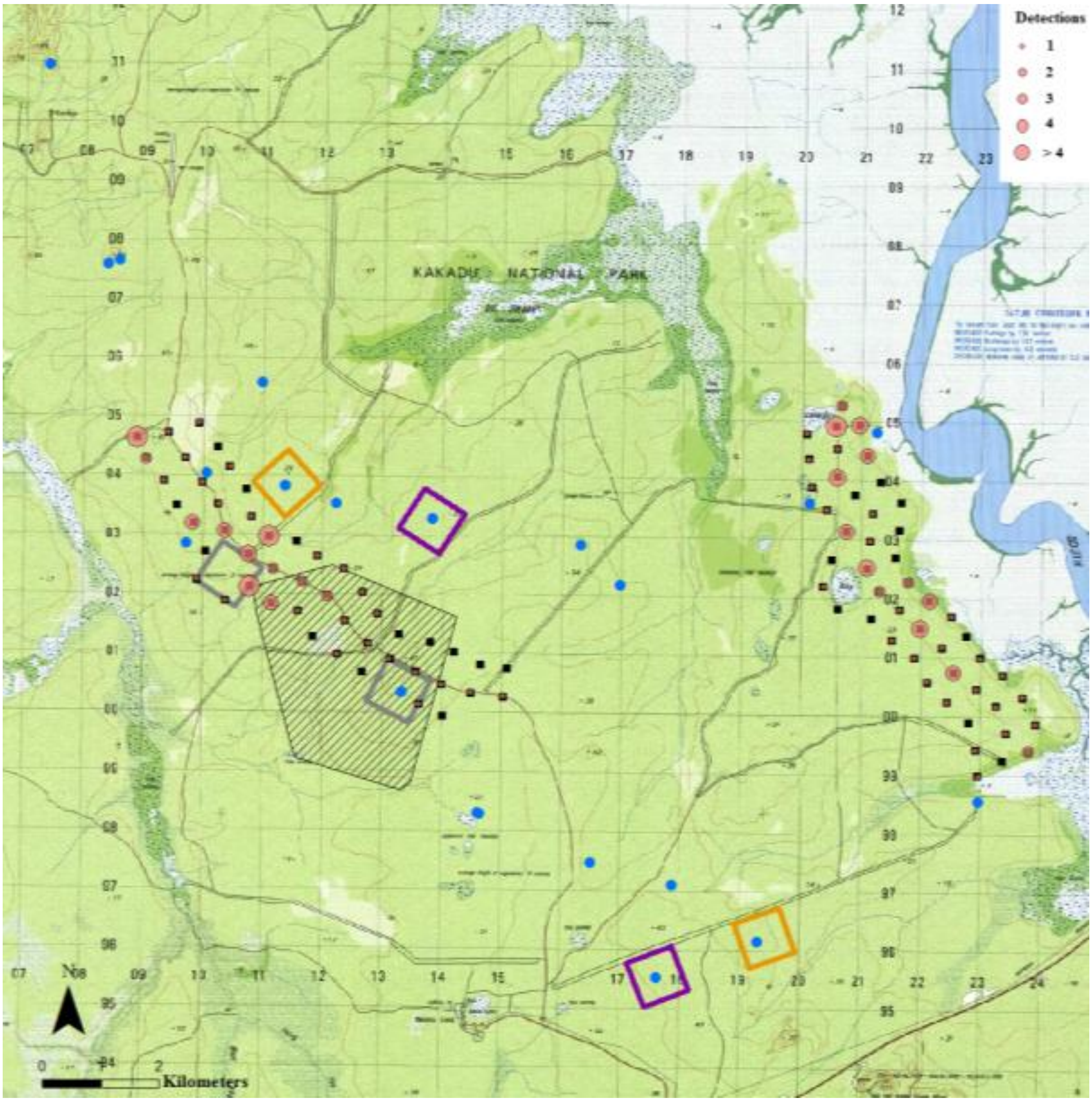


Figure 22. Observations of cats at Kapalga since November 2012 obtained from camera trapping (blue circles). Hatched polygon represents the area used by the cat tracked between Nov 2013 and Jan 2014. Black squares represent locations of the two grids of camera traps deployed in June 2015 for cat density estimates overlayed with the number of cat detections per location (red circles, size proportion to the number of detections).

4.2.3 Predator Diet Analysis

Across the five sampling periods 84 cat and 1100 dog scats were collected and analysed. Single prey species were identified in 97 % of dog scats and 89 % of cat scats, two prey species identified in 3 % of dog scats and 9 % of cat scats, and in 1 % of cat scats three species were identified. Eleven species of native small and medium-sized mammal (11 in cat scats, 6 in dog scats) were identified in the scats, including several species not detected during trapping surveys (Table 12). Pianka's index of dietary overlap was 0.27 suggesting there is only slight overlap in the composition of the diets of dingoes and cats at Kapalga.

Large macropods were the dominant prey item (66 %) present in the dog scats and medium-sized mammals were present in 6 % (Figure 23). In comparison, 44 % of cat scats contained medium-sized mammals, and 21 % contained small-sized mammals. Bandicoot in particular were targeted by both cats and dingoes, ranking as the most frequent species in cat scats and the fourth most frequent in dingo scats (Table 12).

Table 12. Frequency of occurrence (as a proportion of scats) of prey species identified in dog and cat scats collected between June 2014 and November 2015.

Category	Prey item	Cat	Dog
Small Native Mammal	*Leggadina lakedownensis*	1.2	0.0
	Melomys burtoni	3.6	0.4
	Petaurus breviceps	13.1	0.5
	Pseudomys calabyi	1.2	0.0
	Pseudomys sp.	1.2	0.0
	Sminthopsis bindi[#]	1.2	0.0
	Zyzomys sp[#]	1.2	0.0
	Rodent, unidentified	1.2	0.1
Medium Native Mammal	*Dasyurus hallucatus**	3.6	0.0
	Isoodon macrourus	29.8	3.5
	*Mesembriomys gouldii^*	3.6	0.0
	Pteropus Alecto	0.0	0.3
	Tachyglossus aculeatus	0.0	0.9
	Trichosurus vulpecula	11.9	1.7
	Mammal, unidentified	0.0	0.1
Macropod	*Macropus agilis*	7.1	37.5
	Macropus antilopinus	0.0	16.8
	Macropus bernardus[#]	0.0	0.4
	Macropus robustus	0.0	8.2
	Macropus sp.	1.2	4.6
Feral Herbivore	*Bubalus bubalis*	0.0	0.2
	Bos Taurus	0.0	1.9
	Sus scrofa	0.0	2.3
Bird	Unidentified	16.7	3.6
Invertebrate	Unidentified	7.1	0.3
Reptile	Unidentified	4.8	0.1
Other	*Felis catus*	0.0	0.4
	Canis lupus	0.0	1.5
	Human	0.0	0.1
	Bone fragments, unidentified species	2.4	17.3

[#] Species which have not historically been detected in the Kapalga area
* Endangered, *EPBC* Act
^ Vulnerable, *EPBC* Act

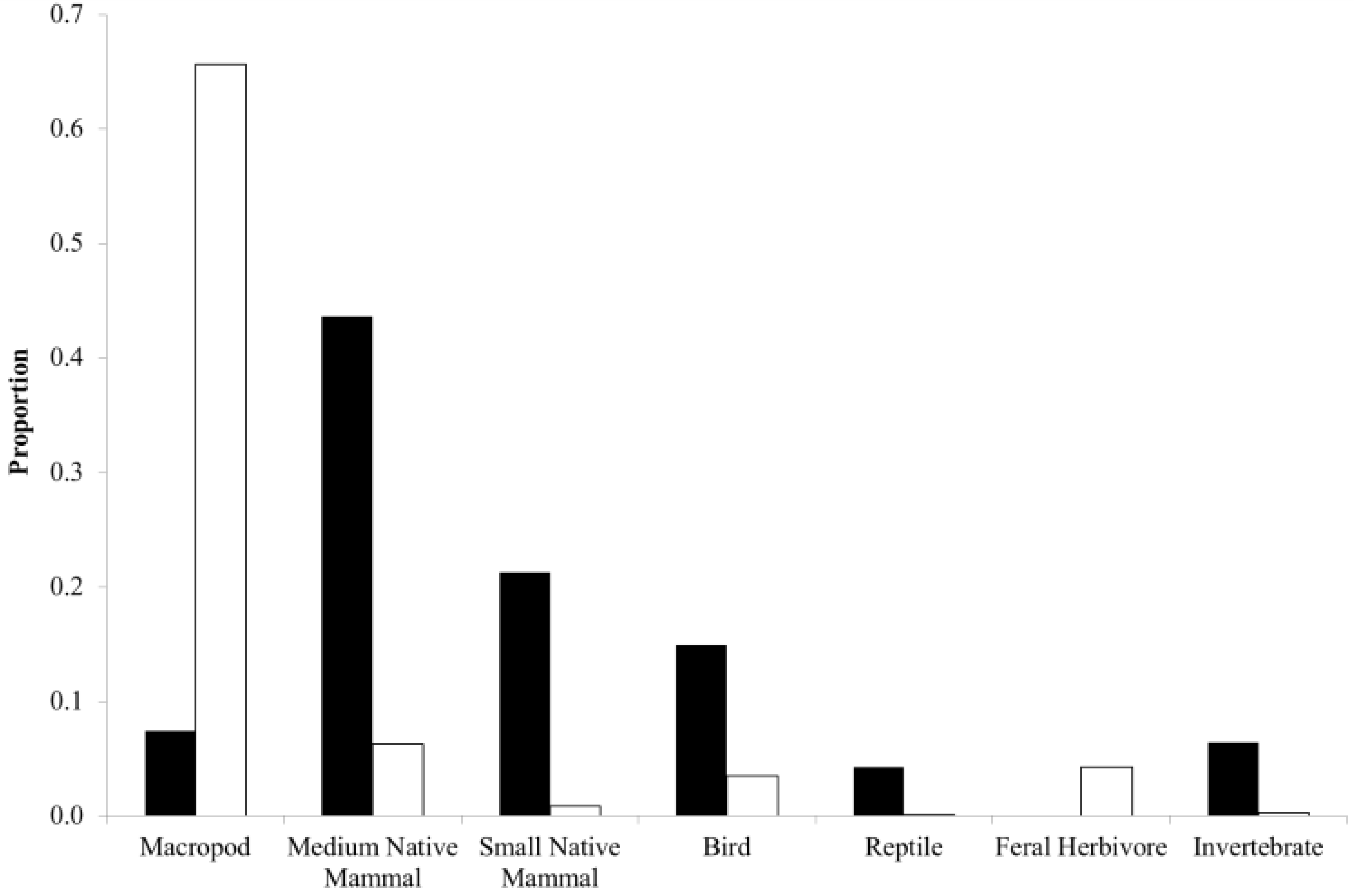


Figure 23. Proportion of prey by category present in dog and cat scats collected across five sampling periods between June 2014 and Oct 2015. Solid bars represent cat scats (n = 84), open bars represent dog scats (n = 1100).

5 DISCUSSION

5.1 Experimental Findings

We found no evidence of population responses of small mammals to the exclusion of feral cats in this experiment. However, the numbers of detections of small mammals across all sites (and through time) were too sparse for any meaningful statistical analyses and interpretation. Therefore these results cannot be considered conclusive for small mammals. These findings contrast with those from the enclosure experiments conducted at Wongalara Sanctuary where rodents were translocated (Frank *et al.* 2014; Tuft *et al.* 2014). There are several possible explanations:

- This result could reflect that cat predation is simply not a major suppressive force on mammals in this region. However, this seems unlikely. Small and medium-sized native mammals were present in over 70 % of the cat scats collected at Kapalga. Furthermore, we obtained evidence of cats from the South Alligator region, and elsewhere in KNP, preying on both small and medium-sized mammals. Of 15 cats dissected from KNP, 11 had remains of mammals in their stomachs including Grassland Melomys, Fawn Antechinus, Sugar Glider, Northern Brown Bandicoot, *Rattus* sp., and *Planigale* sp. (DENR, unpublished data). These preliminary results indicate that, as elsewhere, small and medium-sized native mammals are significant prey in the diet of cats in KNP; however this observation in itself is not direct evidence that cats are suppressing populations of these species.
- Due to constraints on fence design and placement, the exclosures were constructed on flat high ground in open forest, away from drainage depressions, which is not necessarily highly suitable habitat for a broad range of small mammal species. Furthermore, the Kapalga area has sustained high levels of disturbance from intense and frequent fire in recent decades, and introduced livestock. Consequently the inherent small mammal carrying capacity of the experimental areas may be very low.
- Densities of small and medium-sized mammals, for which a numerical response might have been expected, were very low at the commencement of the experiment. This factor, combined with the relatively small exclosure size and short experimental time-frame, may have limited our ability to detect change. Sexual maturity in most dasyurids, likely to persist at Kapalga, is generally reached after eight months (see Jackson 2007); and native rats/mice reach sexual maturity after three months and have small litter sizes, Dusky rats the exception (see Breed and Ford 2007). Therefore, the expected rate of population increases, in cat-susceptible native mammal species, after only two years would not be particularly high. Furthermore, some of the cat-susceptible species, notably the Brush-tailed Rabbit-rat, have not been detected in the study area since 1985 and were likely extirpated from the study area in the last 30 years. Hence no response from this study could be expected in these species as they are simply no longer present in this landscape.
- As a result of the above factors, detection of a measurable numerical response from small mammals in this experiment would have been mostly reliant upon colonisation from adjacent areas, such as monsoon rainforests and floodplains, with residual mammal diversity. Most of the mammal species expected to be present still persist in the general area. However, feral cats (and other predators) are also widespread in the area and may be a significant impediment to such colonisation.

In contrast to small mammals, experimental cat exclusion resulted in a significant increase in reptile diversity, as measured by relative abundance and species richness. Most of the reptile species sampled are expected to have a one year generation time and, based upon the assumption that trap returns are a reasonable index of relative population density, the change in abundance observed within fences represents a generalised reptile species population rate increase of approximately 2, relative to the control site rate of increase. This finding indicates that predation by feral cats is exerting considerable pressure on small reptile populations and diversity.

This finding contrasts with findings from a predator-exclusion experiment at Arid Recovery Reserve in South Australia, which observed a greater abundance of reptiles *outside* predator exclusion areas driven largely by a decrease in gecko abundance within areas where feral predators were excluded (Moseby *et al.* 2009). However, the response of reptiles in arid landscapes might be expected to differ from those in savanna woodland landscapes, due to different top-down and bottom-up ecological processes.

Reptile abundance also increased with time since fire independent of the effects of cat exclusion. Previous studies in savanna woodlands of the Top End have observed short-term responses of reptiles to fire, with decreases in the abundance of some species immediately following burning (Trainor and Woinarski 1994); however Nicholson *et al.* (2006) found no change in the abundance and composition of skinks following low-intensity early-season burns. The response of reptile guilds to fire frequency and intensity is likely to vary based upon habitat use, activity patterns, and life history attributes and warrants further evaluation.

Detection of the threatened Yellow-snouted Gecko on a site increased with time since fire. However, detection in our analysis was confounded with site-occupancy as the population would not have been closed to colonisation and extinction processes between sampling sessions. Previous surveys have observed a greater abundance of individuals at sites unburnt for five or more years (Johansen 2006). This species has a restricted range and is generally found in areas with well-developed leaf litter on red sandy soils (Johansen 2006). It was last recorded within 1 km of one of the study sites (KAP06) in 1988. Although our findings may be supporting evidence that fire is a threat to the Yellow-snouted Gecko, we have evidence that it continues to persist within Kapalga despite the current unfavourable fire regime.

Collectively these findings indicate that both the current prevailing woodland fire regimes and cat predation are having a significant adverse effect on reptile diversity, even though this may not (yet) be apparent in long-term monitoring data or manifesting in increases in threatened species listings.

5.2 Status of Small and Medium-sized Mammals at Kapalga

Capture rates of small mammals were very low across all of the study sites throughout the study period, with some variability between seasons. Elsewhere in KNP small mammals are persisting at low densities, with captures per 100 trap-nights at long-term monitoring sites sampled from 2012 to 2014 also low (mean = 2.0, SE = 0.35, n = 149 sites) (DENR unpublished data).

The use of multiple sampling methods was important for maximising the range of mammal species detected during the study. Although many small and medium-sized mammal species were detected using multiple techniques, such as live-trapping and camera traps, other species were detected using one method but not another. Five small mammal species were observed in predator scats in this study, but not trapped, and other mammal species not trapped were recorded from cat stomach contents (DENR unpublished data). This finding indicates that some species continue to persist in the general area of Kapalga but may occur at densities below thresholds of detection using conventional sampling methods.

These findings have implications for future surveys and monitoring and evaluation programs for native mammals in the Top End. Conventional wildlife monitoring methods are unlikely to be adequate to evaluate long-term population trends of small mammals while densities remain low. Sampling methods for future mammal surveys and monitoring projects in KNP need to be revised to increase sensitivity. Ideally, a combination of trapping methods, incorporating multiple remote cameras with multiple deployment configurations, needs to be developed to improve detection rates of an adequate range of species, along with optimising season and frequency of sampling.

Significantly, Northern Quolls were detected, either with conventional traps or camera traps (Figure 24), in several parts of the Kapalga area during the study. They were captured at rock outcrop sites to the north and south of the study area, but also in the flat, open forests, and riparian floodplain margins (Figure 25). Quolls were captured during seven sampling sessions, with most captures occurring on one control site (KAP06). This finding, combined with the consistent captures of lactating females and sub-adults, is evidence that quolls are breeding successfully in the Kapalga area.

Cane toads, which were widespread throughout the Kapalga region during the study, are considered to have contributed to the decline of quolls in KNP. Quolls are also vulnerable to predation by various predators including feral cats, dingoes, raptors and large snakes (Oakwood 2000), all of which occur in the Kapalga area. In spite of this, and the frequency of fires in the Kapalga area, quolls are persisting in this area of the Park. In view of the population status of this threatened species in KNP, Kapalga may be an important area for more proactive management of the species.

Figure 24. Female Northern Quoll with dependent young captured on camera trap at Kapalga.

Figure 25. Observations of Northern Quoll at Kapalga since November 2012 obtained from camera trapping (blue circles) and live-trapping (black triangles).

5.3 Predator Ecology

5.3.1 Diet

Although frequency methods cannot be used alone to assess carnivore diets, as they may underestimate the importance of large prey items, it is useful in understanding a carnivore's role as a specialist or generalist (Klare *et al.* 2011). Our analyses of dog and cat scats indicated that there is some dietary overlap in the predation of medium-sized mammals. Some dietary partitioning was apparent with medium-sized mammals being the dominant prey group in cat scats whereas dogs preyed more often on large macropods. Cats also took a greater diversity of small-sized animals including birds, reptiles, insects and small mammals. These findings are broadly consistent with other dietary studies of feral cats (Doherty *et al.* 2015b), with the notable exception of the high frequency of bandicoots. This finding may reflect a greater relative availability of this species in the study area; however, this was not reflected in our sampling data – bandicoots were rarely detected, and detections declined during the course of the study.

In a similar study undertaken in the same area of Kapalga in the early 1980s, the dominant prey items in dog scats was native small mammals (34 % of scats) (Figure 26), the bulk of which were Dusky Rat (Corbett 1989). Furthermore, 13 % of the scats contained large macropods, 32 % birds (almost exclusively Magpie Goose), and 12 % medium-sized mammals. Species detected in scats represent a sample from a suite of species available to the predator in a particular time and place. Dietary shifts are likely to be representative of changes in prey availability as dingos generally switch prey as their preferred prey species become unavailable (Corbett and Newsome 1987; Allen *et al.* 2012).

Current and historic wet season population data for Magpie geese are highly variable due to marked, rainfall-driven fluctuations of recruitment rates (Delaney *et al.* 2009). Anecdotally, reports suggest that the northern Kakadu region is no longer an important dry season area for residency by geese as it was in the late 80s, but dry season surveys are not undertaken and therefore there are no data to evaluate this (pers. comm. Keith Saalfield, DENR). Change in frequency of geese in dingo scats may be attributed to this (reported) reduction in dry season residency when dingoes would have access to the floodplain.

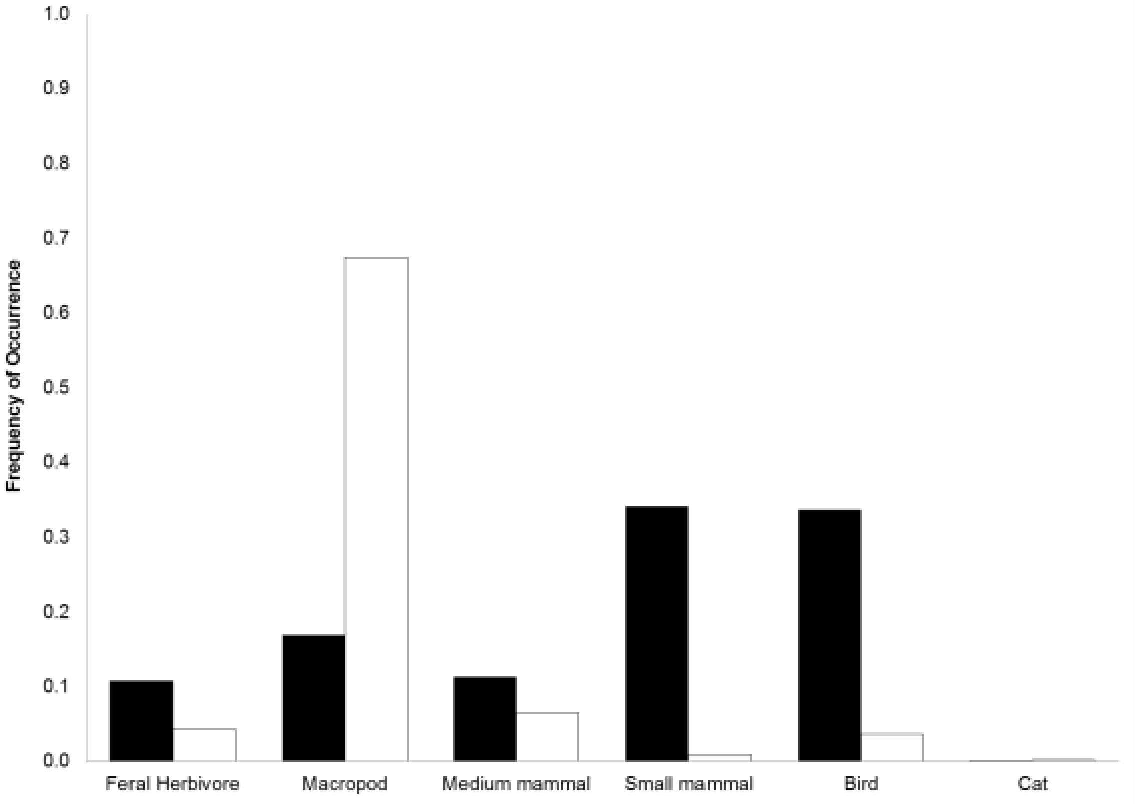


Figure 26. Comparative proportions of prey, by category, present in dog scats collected at Kapalga by Corbett (1989) (solid bars, n = 2495) and the present study (open bars, n = 1100).

Historical cat scat data obtained from Kapalga between 1980-86 (Corbett 1995), also demonstrates the importance of small mammals in cat diets during the 1980s (Figure 27). In the early 1980s small mammal capture rates were much greater than have been observed in more recent times (Figure 28). The observed large frequency shift of small mammals in dog and cat scats between the early 1980s and our recent findings is consistent with the large changes observed in relative abundance of these mammal species in the Kapalga area during the same time period. In contrast, despite substantial declines of medium-sized mammals also occurring during the same time period, the frequency of this group in dog scats dropped by only 50 % and increased in cat scats by 300 %. This finding suggests that the predatory impact of both cats and dingoes may be important for medium-sized mammals. Although currently macropods appear to be the main prey of dingoes they also appear to be preferentially preying upon medium-sized mammals to some extent irrespective of their relative availability.

Our results support the findings of other studies that cats are highly effective predators of small mammals and are able to seek out and prey on them even when their population densities may be very low (reviewed in Moseby *et al.* 2015). However, small mammals also comprised a significant portion of dingo diets when small mammals were more abundant, such as when Dusky Rats undergo population irruptions. Under certain conditions dingoes also have the potential to exterminate or suppress local populations of mammalian species (Allen and Leung 2012). Therefore the predatory role of both cat and dingo populations has implications for the recovery of native mammal populations in the Top End.

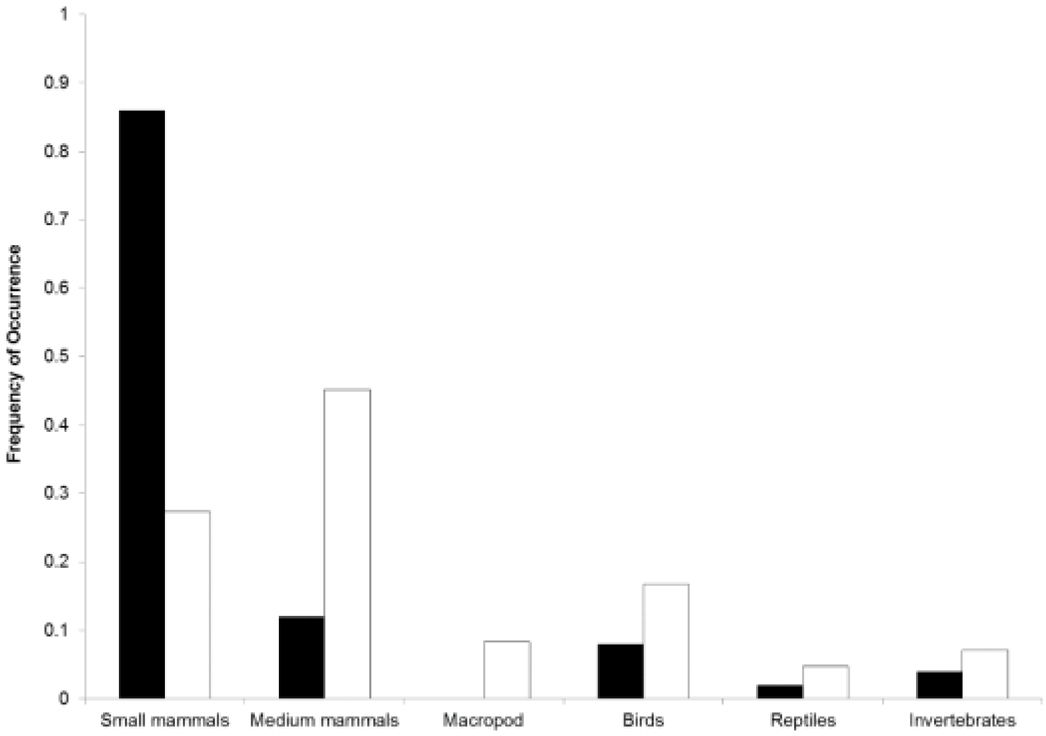


Figure 27. Comparative proportions of prey, by category, present in cat scats collected at Kapalga by Corbett (1995) (solid bars, n = 49) and the present study (open bars, n = 84).

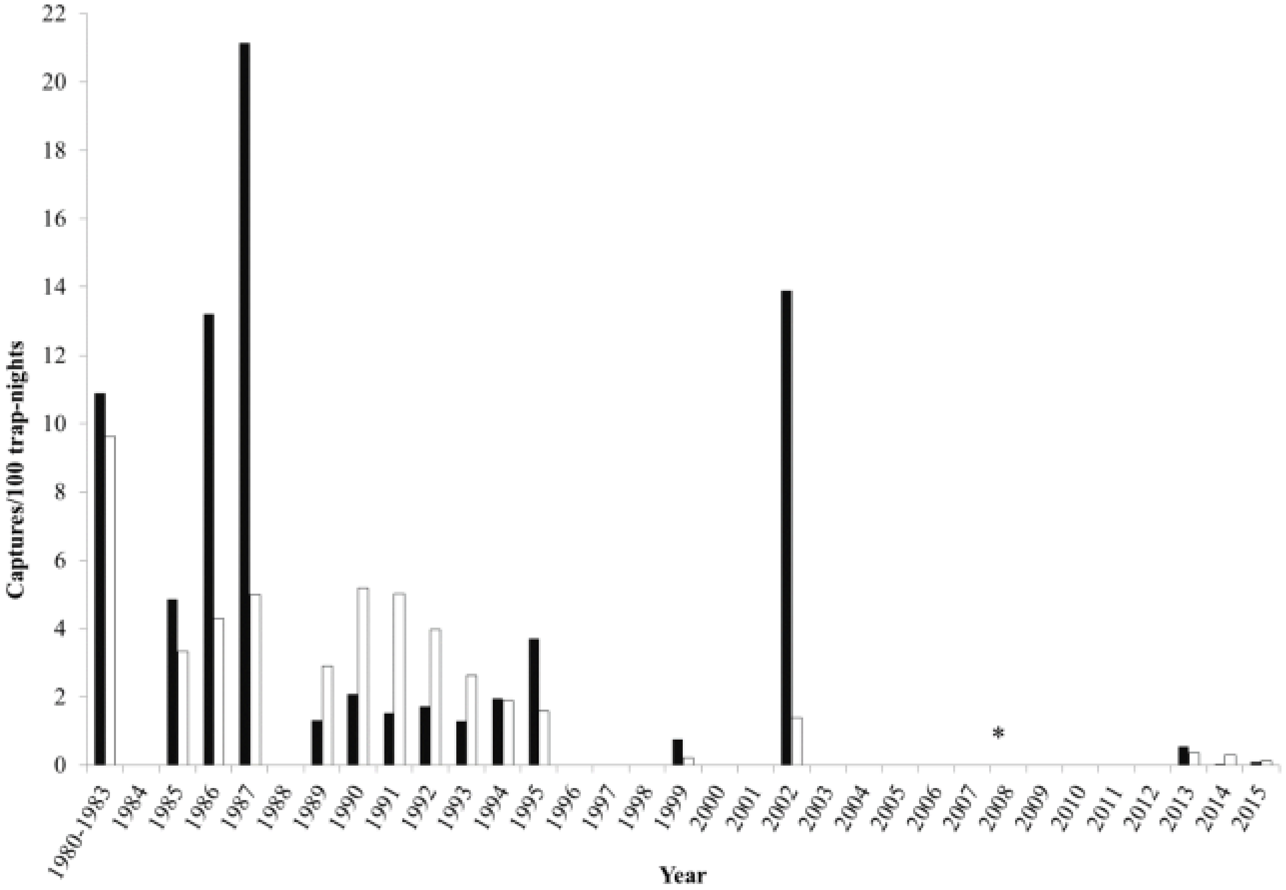


Figure 28. Change over time in capture rate (per 100 trap-nights) in small-sized mammals (solid bars) and medium-sized mammals (open bars). Asterisk indicates no captures were obtained within the year of sampling, and there was no sampling in the years with no bars or asterisk. The spike in small mammal capture rates in 2002 are a result of a high incidence of Dusky Rat captures at one site adjacent to the floodplain.

5.3.2 Cat population estimate

Feral cats were detected by camera and by the presence of their scats throughout the Kapalga study area. They were frequently observed around the perimeters of the experimental exclosures. These observations suggest that feral cats are widespread throughout the region.

Although only a single estimate of feral cat population density was derived during this study, this is the first robust estimate for KNP. The area used by the collared cat and population density estimate are both comparable with estimates for feral cats in the savanna woodlands of the Kimberley (mean home range (male): 855 ha, mean density 0.18 km^2; McGregor *et al.* 2015b). In contrast, home ranges derived for two feral cats in the stone country of western Arnhem Land concurrent with this study, were substantially larger (1563 ha and 2326 ha; DENR unpublished data) than observed elsewhere in northern Australia (H. McGregor *pers. comm.*). A similar camera trap survey from the stone country in the Mary River District of KNP found a cat density of 0.15 km^2 (± 0.05 s.e.) (DENR unpublished data). Although very preliminary, these estimates suggest that there are approximately 3,300 (CI: 1518 – 6173) cats in KNP.

Both bottom-up and top-down processes, including prey availability and competition with sympatric predators, are likely to influence the distribution and density of cats (reviewed in Doherty *et al.* 2015a). Gaining a better understanding of the ecological factors that influence and regulate cat populations in tropical savannas will be important in developing and implementing management strategies to mitigate biodiversity impacts of feral cats.

5.4 Study Limitations

There were a number of difficulties encountered in the planning and implementation of this project which affected the outcome of the research and the conclusions that can be drawn:

The original intent for this experimental research project was for fauna sampling to be undertaken for a three year period following fence construction. However, due to significant delays experienced in the consultation process and approvals for site selection, responses of fauna to the treatments were only measured for two years. This time-frame may be inadequate to observe changes in mammal diversity as a result of experimental manipulation to exclude cats. Although some small mammals persist in the study area at low densities, more time may be required for populations to reach densities whereby significant changes in their populations can be detected. In a predator-proof reserve in arid South Australia, a population response was not observed in rodent species until four years after predator removal (Moseby *et al.* 2009).

The initial project plan was to replicate the experiment at two different locations; Kakadu National Park and either Litchfield National Park, Garig Gunak Barlu (Cobourg) National Park or Bathurst Island. However, a combination of financial constraints and lack of land manager support precluded experimental replication. The strength of experimental replication is the ability to gain confidence and draw general inference from the conclusions. Because biological systems are inherently variable, the aim of replication is to compare the variability between the treatment groups to the inherent variability between the experimental units (sites) in each group. The absence of such replication in different environments limits what general inference can be drawn from the findings of this research.

Fire is an important driver of habitat change in tropical savannas (Williams *et al.* 1999). Recent studies by the Australian Wildlife Conservancy have demonstrated that cats preferentially hunt along fire scars, where reduction in the understorey vegetation is likely to make small mammals easier prey for cats following fire (McGregor *et al.* 2014). It was therefore desirable to control for, or compartmentalise, the effects of fire in this experiment to minimise the risk of confounding interpretation of the predator treatment effects. However, fire of varying intensity and extent occurred at all six experimental sites both before and during the experiment (2013 – 2015). Both fenced sites and one firebreak site were burnt just prior to fence completion. Intensive and careful management of fire in this region will be critical for any future utility of the experimental infrastructure at Kapalga.

The following methodological limitations in the study design may confound the interpretation of results from this experiment.

1. In the event that a measurable response of small mammals had been observed in the experimental exclosures, assigning this change exclusively to predation by cats would be problematic. Available evidence indicates that both cats and dogs prey on small and medium-sized mammal species to varying degrees, and the fences exclude both predators. Both species occur throughout the Kapalga area.
2. Some natural variability is expected between experimental units. However, inherent changes between experimental treatments are compounded through the exclusion of feral herbivores (including pigs) and fire from the fenced sites. Both fire and herbivory impact on habitat structure and vegetation composition. The response of small vertebrates to changes in habitat structure is complex and expected to vary amongst species. Some changes in small vertebrate diversity are likely to result from changes in habitat structure alone.

Because of the issues outlined above, it may be difficult to confidently identify a single mechanism, such as cat predation, underpinning faunal responses in this experimental design. If possible and practical, future studies of this type should consider designs that minimise the confounding effects of multiple ecological factors.

6 CONCLUSIONS AND MANAGEMENT IMPLICATIONS

6.1 Exclusion Experiment

The cat exclusion experiment was inconclusive in assessing the impacts of feral cat predation on small mammal populations in KNP. There are several plausible explanations for the results, outlined in Section 5.1.

The experiment demonstrated that predation by feral cats is exerting considerable pressure on small reptile populations and diversity in savanna woodland habitat. Albeit to a lesser degree, frequent fire is also exerting suppressive effects on small reptile populations. The effects on reptile populations may not yet be apparent in long term monitoring data or resulting in increases in threatened species listings. The findings may be an early warning for species such as the Yellow-snouted Gecko, which may be suffering declines that have not been detected with current monitoring approaches.

Further monitoring of reptiles at these experimental plots would yield important information on the relative vulnerability of different components of the reptile community, and help to hone management priorities. Furthermore, evaluation of guild and individual species responses to cat predation, fire and their interactive effects could provide insights into how these processes are re-organising reptile communities, and help inform optimum fire regimes for maximising savanna reptile diversity.

6.2 Predation and Mammal Declines

In spite of the inconclusive results for small mammals in the predator exclusion experiment, collectively our findings provide further evidence that feral cats are having a significant predatory impact on vertebrate fauna in KNP. These results build on other research undertaken elsewhere in Northern Australia and collectively suggest feral cat predation is playing a significant role in mammal declines (Frank *et al.* 2014; Tuft *et al.* 2014; Leahy *et al.* 2016). Despite the now exceedingly low densities of most small and medium-sized mammals in KNP, they continue to be significant component in the diet of cats. In addition, there were demonstrable impacts of cat predation on small reptiles, and recent camera trap surveys suggest that feral cats are widespread throughout the Park (DENR unpublished data).

Predation by feral cats has been identified as the likely primary proximal cause of northern Australian mammal declines (Woinarski *et al.* 2015). Their predatory impact may be exacerbated across northern Australia in recent decades by inappropriate fire regimes and habitat alteration by livestock (McGregor *et al.* 2014; Ziembicki *et al.* 2015; Leahy *et al.* 2016). In particular, burning regimes which remove ground cover can amplify the impacts of predators (Leahy *et al*. 2016). Our findings also suggest that dingoes/dogs may be able to exert significant predatory pressure on some mammal species under certain conditions. Dingo predation, even at low levels, may negatively affect vulnerable prey, such as those affected by other threatening processes (Johnson *et al.* 1989; McCallum *et al.* 1995).

Abundance indices have been used in northern Australia to demonstrate an inverse relationship in dingo and cat densities (Brook *et al.* 2012; Kennedy *et al.* 2012). However, these studies did not consider the issue of species detectability in their analyses which could bias the results. Cats were only infrequently present in dog scats, both historically (< 0.01 % of scats in Corbett's study) and in the current study, despite being present at densities comparable to other areas in northern Australia. Schroeder *et al.* (2015) demonstrated in the arid-zone that dingoes do not actively hunt cats but will opportunistically kill them when given the chance. Generally there is still little known about the ecological interactions between dingoes and cats in northern Australia.

The ecology of cats and dogs in savanna ecosystems, and the interactions between these predators and native mammal population demography, are likely to be complex (Ziembicki *et al*. 2015). Predicting the outcomes of landscape management on these communities is difficult. Current knowledge of population dynamics of dogs and feral cats and their interactive effects in KNP and similar ecosystems is poor. Radford *et al.* (2014) presented a conceptual model of how ecosystem productivity can influence predator interactions and regulate mammal diversity in northern Australian savanna ecosystems. They stress the importance of careful, multi-faceted monitoring and evaluation of any management interventions. Management actions to address mammal declines that focus on ameliorating the impacts of fire and other disturbance processes are a logical next step in areas such as KNP. However, consideration and evaluation of as many factors as possible that may be influencing outcomes for mammal diversity (and small vertebrates generally), will be important to improving our understanding of underlying mechanisms and to further refine management actions in the future.

6.3 Mammal Diversity of Kapalga

The use of a wide range of sampling methods, including conventional live-trapping, camera traps, scat and gut analyses demonstrated that many native mammal species continue to persist in the Kapalga area but at densities below thresholds of detection using conventional trapping. These findings suggest that much residual mammal diversity persists in the Park, providing a potential base for the recovery of populations if appropriate management regimes can be developed and applied. However, no evidence of persistence was obtained for Brush-tailed rabbit-rat *Conilurus penincillatus*, Northern Brush-tailed Phascogale *Phascogale pirate* (Vulnerable, *EPBC* Act), Long-tailed Planigale *Planigale ingrami*, Common Planigale *Planigale maculata*, Red-cheeked Dunnart *Sminthopsis virginiae,* Western Chestnut Mouse *Pseudomys nanus*, or the Pale Field-rat *Rattus tunneyi*.

For most mammal species that can still be detected with conventional trapping methods, trap-rates are less than 0.5% of those recorded prior to declines. These findings demonstrate that wildlife survey and monitoring methods historically employed in this region are likely to be inadequate for reliably estimating current patterns of mammal persistence, or evaluating temporal changes in species' populations or assemblages. Advances in camera trap methods have partly ameliorated this limitation (Gillespie *et al.* 2015), but further developments and refinements are needed to improve sensitivity for small-sized mammals.

Recent surveys and monitoring in KNP and surrounding areas indicate that quolls are either absent or sparsely distributed throughout the region. An extant population of quoll is known near the East Alligator Ranger Station (Cremona 2015); however, this population was severely predated by wild dogs in recent years. Recent extensive camera-trapping in sandstone escarpment habitat near the Mary River Ranger Station, and at fire plot sites in the general vicinity, failed to detect any quolls (DENR unpublished data). Extensive camera-trapping and live-trapping in Nitmuluk National Park, Warddeken and Djelk IPAs since 2012 also failed to detect any quolls (DENR unpublished data). The only quolls detected during the most recent sampling round of 134 fire-plot monitoring sites in KNP were in the Kapalga area. In this context, the quoll population in the Kapalga area is quite unusual, and perhaps unexpected, given the suite of pressures operating on the species. In view of the threatened status of this species and the importance placed upon its management and recovery in KNP, more proactive interventional management of the threats operating on quolls in the Kapalga area should be considered, along with appropriate monitoring and evaluation of the population.

6.4 Predator-Proof Fences

There is currently much interest in the efficacy of predator-proof fences for biodiversity conservation in Australia. Circumstances exist where fences may be considered the most effective means for the protection of critical populations of threatened species, or their potential re-introduction. This project was the first time that predator-proof fences have been constructed and maintained in the wet-dry tropics. Although relatively small-scale compared to fences employed for long-term conservation initiatives, the learnings gained from this project should be useful in informing and evaluating the efficacy of predator-proof fences as a management tool elsewhere in similar environments (Appendix I).

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APPENDIX I: FENCE CONSTRUCTION AND MAINTENANCE COSTS

The combined costs of vegetation clearance for fire easement and fence construction were approximately $65,000 per km. The per-km cost of fence construction may be lower for a longer, single, continuous fence; however, a permanent fence for conservation management purposes would require a wider easement. We employed an 8 m cleared easement, which meant that large branches and whole trees could still severely damage the fence (and did so on several occasions). In woodlands with canopies of approximately 20 m, easements would need to be at least double this width to significantly minimise the risk of falling trees. However, this would not preclude ongoing fence maintenance to address any vegetation regrowth along easements, including weed establishment, or to identify trees that may risk fence integrity in the future. Furthermore, as witnessed at Kapalga, large feral herbivores are equally capable of causing significant damage to these fences.

Fire is a persistent issue in the savanna woodlands, and even with wide easements leaf litter will accumulate along the fence apron over time which can ignite from windblown embers. Fire can accelerate peeling of the zinc-iron alloy layers of galvanised steel which will result in reduced corrosion protection. Rusting of the foot apron is likely to occur over time and will need to be replaced as rusting is observed. Rust patches developed in the fence at the Arid Recovery Reserve eight years after construction where water consistently pooled following rain (B. Hill *pers. obs.*). In the tropical north, it is prudent to assume that rusting may occur more quickly than in the arid centre. Preparing the ground to ensure proper drainage along the fence-line will be important to prevent water from pooling. However, replacement of the foot apron will likely need to be incorporated into on-going maintenance costs in the longer term. For any predator-proof fences planned for biodiversity conservation, general fence maintenance and hardware replacement costs will need to be pre-empted and budgeted for in project costs.

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