ECOLOGY OF THE FERAL CAT (*FELIS CATUS*) IN THE TALL FORESTS OF FAR EAST GIPPSLAND

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Declaration

I hereby declare that this thesis is my own original work and that it contains no material previously published or written by another person, except where due acknowledgement is made.

Signed: ....

Anthony John Buckmaster

Dated: ....

The gods were once disputing whether it was possible for a living being to change its nature. Jupiter said "Yes," but Venus said "No." So, to try the question, Jupiter turned a Cat into a Maiden, and gave her to a young man for a wife. The wedding was duly performed and the young couple sat down to the wedding-feast. "See," said Jupiter to Venus, "how becomingly she behaves. Who could tell that yesterday she was but a Cat? Surely her nature is changed?"

"Wait a minute," replied Venus, and let loose a mouse into the room. No sooner did the bride see this than she jumped up from her seat and tried to pounce upon the mouse. "Ah, you see," said Venus, "Nature will out."

Aesop (620 – 584 BC)

"The cat is relatively easy to house, feed and handle, but despite the process of domestication it has retained a behavioural repertoire which makes it very successful in the feral state..."

Bradshaw *et al.* (1996)

Abstract

The house cat, *Felis catus*, was introduced into Australia with European settlement of the mainland. Since its initial introduction, it now occupies all mainland habitats, Tasmania and many smaller offshore islands. Large numbers of cats were released intentionally into the environment in a misguided attempt to control the spread of other introduced mammalian pests, especially the European rabbit, *Oryctolagus cuniculus*. The feral cat is an invasive predator that has been implicated in the decline and extinction of many species of native small mammals across Australia, particularly in the arid regions and on offshore islands. Much of the research on feral cats in Australia has occurred in the continent's arid and semi-arid regions. Consequently, little is known about the ecology of feral cats in tall forests. Additionally, the most generally effective population control technique, poison baiting with sodium monofluoroacetate (compound 1080), has wide ranging applicability in arid and semi arid areas but its use is restricted in the temperate and forested eastern states of Australia due to concerns about impacts on non-target species.

This thesis is divided into three parts. Firstly, I review the current knowledge of feral cats, particularly in relation to the actual and potential impact they have on native prey species. Secondly, I investigate the ecology of the feral cat in the temperate tall forests of Far East Gippsland, Victoria. The home range sizes, movement patterns and home range use of feral cats were determined. Thirdly, I examine a new technique for delivering poisons in a feral cat management program. The potential for all Australian non-target species to access the toxicant is examined using a desktop analysis, while field studies examine uptake by non-target species and the dynamics of prey species to determine acceptable times for baiting campaigns.

GPS and VHF collars were utilised to obtain fix data for feral cats in Far East Gippsland. Male cats had significantly larger home ranges (MCP100 455 ± 126 ha) than females (105 ± 28 ha), with male home ranges overlapping those of females. Some female home ranges overlapped extensively, with neighbouring females also having overlapping core areas within their ranges. These overlaps in female home ranges, in particular of the core areas, indicate that female cats in Far East Gippsland are tolerant of other females and do not actively exclude them.

Compared with the home ranges of feral cats in other regions of Australia and New Zealand, the cats in Far East Gippsland had smaller home ranges than those of cats occupying arid and alpine zones yet larger ranges than those of feral cats living in farmland or grassland. This variation probably reflects the availability of food resources, with cats in resource-poor areas requiring larger home ranges and cats with smaller home ranges generally inhabiting areas with greater, or more accessible, food resources.

The use of GPS collars to obtain accurate and high volumes of location data allowed the intra-home range movements of feral cats to be examined in ways not previously possible using conventional VHF radio telemetry. Location data were gathered at three different temporal intervals – 6 hourly, hourly and every 15 minutes. Feral cats followed a Lévy walk-style searching pattern as they moved through their home range. Employing a Lévy walk increases the likelihood of encountering prey items that are distributed sparsely in the environment, in turn maximising the potential hunting return for effort expended.

Each of the cats examined had large areas within their home range that they did not enter. To test the hypothesis that this resulted from a scarcity of prey in these areas, trapping grids were established to capture small prey-sized animals. There was no difference in the rate of capture of prey species in the areas of high and zero cat use, thus allowing the food hypothesis to be rejected. Modelling of abiotic environmental parameters was used to determine if these influence home range use. While the models explained much of the variation in the data, the global model was overdispersed, indicating that other unmeasured parameters were influencing home range use. The avoidance of these areas most likely arises from the presence of larger intraguild predators and subsequent employment of predator avoidance strategies by the cats.

Managing the abundance of feral cats using poison baiting requires that bait be distributed at times when cats are food-stressed. Generally this occurs in winter when prey species are in natural decline. To determine the most appropriate time for baiting feral cats, trapping grids were established to assess the population demographics of feral cat prey species. The 2 046 trap nights undertaken resulted in 176 captures of five prey-sized species. The breeding periods for the *Antechinus* spp. occur earlier in Far East Gippsland than would generally be expected based on the latitude and altitude of the trap sites, and have bearing on the optimal time for poison baiting. Based on these findings, the optimal time to manage feral cat populations in Far East Gippsland through poison baiting is between late August and mid November provided that the toxicant is enclosed within a hard shell delivery vehicle (HSDV) that maintains structural integrity or, alternatively, if the baits are suspended above the ground surface and out of reach of lactating female antechinus. Further research is proposed to supplement these findings.

Encapsulation of toxicants within an acid soluble HSDV which is then inserted into the bait media is being explored as a potential technique to minimise access of non-target species to the toxicant. A desktop analysis employing a decision tree process was used to examine the potential for non-target access to toxicant delivered in an HSDV. This analysis encompassed all non-aquatic vertebrate species in Australia. significantly fewer species would be susceptible to non-target poisoning if HSDVs were used when compared with directly injecting the toxicant into the bait media. Carnivorous mammals were the most likely to consume both the bait and the HSDV.

Using the systemic marker, Rhodamine B (Rb), in the HSDV, I assessed the ability of five species of small to mid-sized animals to access toxicants enclosed in the HSDV. This was compared with directly injecting it into the baits. Rhodamine B staining was apparent in the mystacial vibrissae of four of the five species at sites where Rb was injected into the baits. It was also present in three of the four species captured at the sites where the Rb was encapsulated within the HSDV. The longevity of the HSDV within the bait media was tested and found to decreased rapidly following insertion into the bait. This is most likely due to the bait media being slightly acidic. Since that experiment concluded, changes have been made to the pH of baits to extend the integrity of the HSDV and hence reduce leakage.

These key findings will allow managers to adopt a more targeted approach when undertaking cat control programs in these habitats. The use of GPS technology to obtain location data has allowed the analysis of intra-home range movements to an extent previously not possible with other techniques. This in turn will allow a more targeted approach to managing feral cats. The use of a decision tree approach to determining the susceptibility of non-target species during a baiting campaign can be applied to other poisoning campaigns regardless of the target species or the toxicant being used.

Acknowledgements

Many people and organisations have contributed time, funding, support, guidance and assistance to this project. I owe them all an enormous debt of gratitude and I sincerely apologise if I have missed anyone.

First and foremost I thank my wife Jo, and children, Alex and Georgia. The gift you have given me in allowing me to chase my dreams is beyond measure. Thank you for tolerating my absences to the field and all parts far away, and for being there with open arms and unconditional love and affection when I return. You have all given so much to allow me to do this and mere words cannot express my gratitude satisfactorily.

I also owe a special debt of gratitude to Chris Dickman, my primary supervisor. His guidance, advice and gentle “have you considered....” have made my research and this thesis far more that it could have ever been without his guiding hand. I completely echo the sentiments of many former doctoral students of Chris in that he is, and always will be, my model as a scientist.

I am very appreciative of the friendship and support of my fellow student, Alex Diment who worked closely with me and helped me with my project even though he was running his own research project at the same time.

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I am sincerely grateful to Dave Algar who saw that I knew very little about catching cats and freely gave his time and advice and showed me all he knew about how to catch cats. All the Eradicat® baits and most of the trapping equipment used in this project were provided by Dave. Without his wise advice and equipment, n=3 cats would have made for a very thin thesis. I am also grateful to Steffi Hilmer lending me an ear, providing encouragement and for the cool souvenirs from her overseas trips.

The Invasive Animals Cooperative Research Centre has been most generous with its funding of both this project, topping up my scholarship and paying my part time scholarship. I thank Tony Peacock and Steve Lapidge for having the enthusiasm and foresight to support a feral cat project and, during the dark days of when nothing was going right, for having enthusiasm for my project when I had none. Thanks to Nina Jenkins and Tom Heinsohn for arranging for my attendance at conferences and training workshops. A special thank you must go to Alex Bagnara, Di Holloway and Sascha Rettke for tolerating my bad jokes, feeding me lollies and allowing me to invade their various offices when I was at a loose end.

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I am sincerely grateful to David Petterson and his children, Billy and Jacqui, for welcoming me into their family, for regularly providing me with hot meals accompanied by red wine and fairly rugged port - and for teaching me how to drive a bulldozer. David is the worst weather forecaster there is, as can be attested to by the times I followed his sage-like weather advice and ended up standing on a bare hilltop in the rain holding an aluminium yagi above my head and watching an approaching thunderstorm. However, he is the best friend I could have asked for, giving freely of his time, knowledge and possessions and never expecting anything in return.

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I also need to sincerely thank my surgeon, Mr Graeme Stewart, for unzipping me, rummaging around and zipping me back up. I am grateful for his tireless efforts to keep me fit and healthy but am sorry that I asked, and was told, what “you may experience some discomfort” actually meant prior to one procedure.

All the research undertaken for this thesis was approved by the Animal Ethics Committee at the University of Sydney (L04/8-2006/3/4457). Field work in Victoria was carried out in accordance with the National Parks Act 1975 and the Wildlife Act 2975 (permit 10003945) and under the Forests Act 1958 (Permit EGFMA-06-01)

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Chapter 1

General introduction and study site description

Introduction

The domestic cat (*Felis catus*) was introduced into mainland Australia with the first European settlers (Abbott 2002). It has since spread across the mainland and to Tasmania and many offshore islands through a combination of intentional releases and natural dispersal (Abbott 2002; Burbidge *et al.* 1997). Feral cats have been implicated in the decline of many populations of native Australian animals, in particular native mammals and birds on offshore islands (Burbidge *et al.* 1997; Dickman 1996; Dickman 2009). Introduced predators can have a far greater impact on native prey populations than predators with which prey species have co-evolved. Naïve prey may not have adequate or appropriate predator avoidance strategies (Banks 1998; Salo *et al.* 2007).

The ecology of the feral cat has been examined to a limited extent in the arid and semi arid regions of Australia (Algar and Burrows 2004; Read and Bowen 2001; Risbey *et al.* 1999) and to a lesser extent still in temperate woodlands (Molsher 1999; Molsher *et al.* 1999). Few studies have been directed towards feral cats in high productivity, tall forest habitats. Dietary analyses were carried out by Friend (1978) and Triggs *et al.* (1984) on cats in the tall forests of Gippsland, Victoria. These are the only specific studies that appear in the literature dealing with any aspect of feral cat biology in this habitat type in Australia, although several wider-ranging studies have presented data on cat diet as part of reviews or broader works on guilds of forest predators (Glen *et al.* 2011; May and Norton 1996).

Overseas research, while more extensive than that in Australia, has been directed mostly towards the ecology of stray cats (Mirmovitch 1995; Natoli *et al.* 2007; Page *et al.* 1992), or feral cats in open habitats such as farmland (Langham 1992; Langham and Porter 1991) or on islands (Apps 1986; Baldwin 1980; Kirkpatrick and Rauzon 1986; Konecny 1987).

In this thesis, I examine the ecology of feral cats in the tall closed forests of Far East Gippsland, Victoria. I describe patterns of movement and spatial behaviour of feral cats within their home ranges, assess possible causes for the patterns observed and examine novel control techniques for the management of feral cats. Some focus is also placed on the responses of prey species of small mammals to feral cats. In this chapter, I briefly introduce the cat and the definitions used throughout the thesis, and describe the tall forest region of Victoria where the study was carried out. Contrary to convention, I defer the primary justification for my work until chapter 2 and note here only that the feral cat is a particularly problematic species with respect to the management of its impacts on native Australian fauna. I adopt this slightly different arrangement to allow for a more thorough review of the ecology of the feral cat than would be usual in an introductory chapter, and then use the review to help inform the direction that my research should take.

Definitions used in this thesis

Felis catus: The domestic cat with a blotched tabby pelage was designated *Felis catus* by Linnaeus in 1758. More recently, phylogenetic studies have attempted to define and place the species within the cat clade. The European wildcat (*F. silvestris*), African wildcat (*F. lybica*) and the domestic cat (*F. catus*) appear to have diverged from their historical lineage as a polytomy, prompting some authors to advocate the classification of these three putative species as subspecies within the single taxonomic concept of *F. silvestris* (Johnson *et al.* 1996; Johnson and O'Brien 1997; Masuda *et al.* 1996; Randi and Ragni 1991). However, until this occurs and formal taxonomic recognition of *F. silvestris* is accorded, *F. catus* remains a species in its own right and shall be called such in this thesis.

Introduced (or invasive) predator: I restrict this definition to only the fox (*Vulpes vulpes*) and the cat in this thesis. The fox was introduced into Australia following European settlement with the first successful release occurring at Geelong, Victoria in 1871 (Rolls 1969; Saunders *et al.* 1995). While there is still robust discussion in the literature about the origins of the cat in Australia, I have accepted the time line proposed by Abbott (2002) that the cat is a European import. The dingo (*Canis lupus dingo*) is excluded from this definition as it has been on the Australian mainland for 3 500 – 4 000 years (Johnson and Wroe 2003) and is now considered by many to be a native predator. This situation stands in marked contrast to the much more recent arrivals of both the cat and fox.

Home Range: Home range is described by Burt (1943) as the area traversed by an animal in the course of its normal activities of gathering food, mating and caring for young and does not include occasional, perhaps exploratory, sallies outside this area. I use Burt's (1943) definition in this thesis. Home ranges of individual cats may overlap with those of other cats. The existence of a home range does not necessarily imply, or preclude, the possibility that all or part of that range is defended by the resident(s) against intraspecific intrusion.

Territory: This is taken to be an area equivalent to all or part of a cat's home range that is defended against intrusion of by other cats, especially cats of the same sex. Territories of male and female cats may overlap, however, with defensive behaviour being triggered only by intrusion of a cat of the same sex as the territory resident.

Classes of cat

Many differing definitions have been formulated to describe the various states of cats with respect to their interactions with humans and with the environment. Many of the categories used by one author overlap with other categories used by another author (e.g. Baker *et al.* 2010). For example, free-living cats defined by Denny *et al.* (2002) are described as stray cats by Say and Pontier (2004) and as semi-feral cats by Liberg *et al.* (2000). For the purposes of continuity, I will use the definitions proposed by Dickman (1996), who followed those of Moodie (1995).

> **Domestic cat**: A cat that is owned and lives in close connection with humans who assume most, if not all, responsibility for providing the obligate requirements of food and shelter. A domestic cat may still hunt for food or for

play purposes. Food obtained through hunting is additional to that provided by the person or persons assuming responsibility for that cat.

Stray cat: A cat that lives in proximity with humans but is not wholly dependent on humans for the provision of food and shelter. They are not owned and no person takes absolute responsibility for all their requirements although they may be fed by humans on a regular basis. These cats are found in towns, garbage dumps and dockyards (Dards 1983; Denny *et al.* 2002; Liberg *et al.* 2000; Page *et al.* 1992). Also included in this category are those cats defined as farm cat by various authors (for example, see Barratt 1999; Hamilton *et al.* 2006; Langham and Porter 1991). Farm cats are cats living on farms where they are allowed to remain by the farmer primarily for the purposes of rodent control. They occasionally may be fed by the farm owner, but generally survive on rodents caught around the farm.

Feral cat: Feral cats are free-living cats that do not rely on humans for any portion of their obligate resource requirements, either directly or indirectly, and have formed self-sustaining populations.

It should be noted that individual cats may move between classes on several occasions throughout their lives. For example, a domestic cat may become stray or feral and *vice versa*. Furthermore, the boundary between each class is not sharply defined. There is no firmly established point along the gradient at which cats change from being wholly feral to wholly domestic. The class boundaries thus are somewhat blurred and based on the circumstances of individual cats at any given time (Dickman 1996).

Study area

Research for this thesis was undertaken in the Far East Gippsland region of Victoria, Australia. This region is characterised by tall forests that span the area from coastline to mountain peaks. Most land in the region is managed by the Victorian Department of Sustainability and Environment (DSE) for either forestry purposes (through the sub-department, VicForests) or for protection as national parks and reserves (through Parks Victoria). However, there are pockets of privately owned land in the region.

These are mainly confined to the southern, lowland river valleys where fertile flat land occurs due to a combination of erosion of higher land and flood deposits. Additional private land is located on the north eastern highland plateaus.

Climate

Far East Gippsland has an overall temperate/mesothermal climate. Cool winters and warm summers prevail. Due to the variety of land forms throughout the region, however, the climate varies dramatically between the coastal plains and the mountain areas or, as characterised by the Köppen climate classification scale (Peel *et al.* 2007), from oceanic to subtropical highland climates. The region receives a mean annual rainfall of 910 mm with rainfall decreasing from east to west and similarly decreasing from the coast towards the inland areas. Monthly rainfall patterns are trimodal with rainfall peaking in May and lesser peaks in November and February (Figure 1). Mean maximum temperatures range from 14.2°C in June to 26.1°C in February. Mean minimum temperatures range from 2.1°C in July to 13.5°C in February (Figure 2) (BoM 2010).

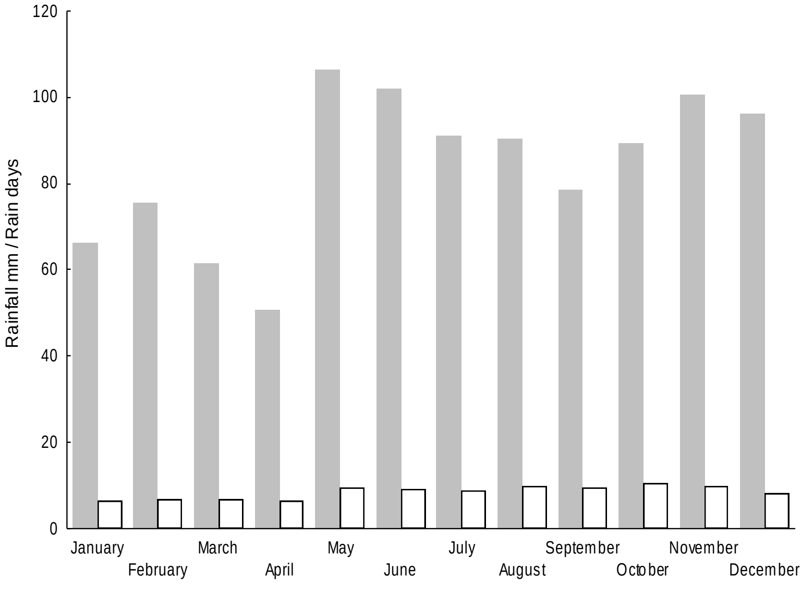


Figure 1. Mean monthly rainfall pattern for Far East Gippsland. Solid bars show mean monthly rainfall. Hollow bars show mean number of rain days per month. Data obtained from the Bureau of Meteorology, Combienbar weather station.

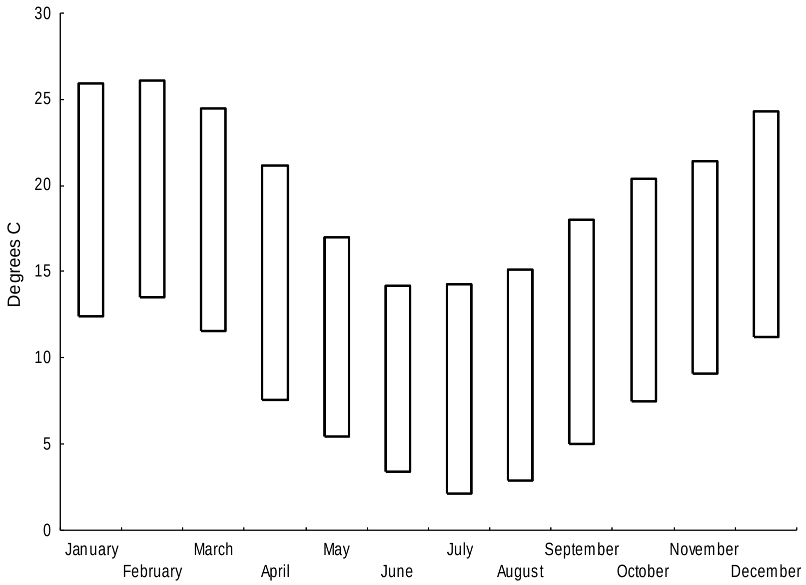


Figure 2. Mean monthly temperature pattern for Far East Gippsland. Bars indicate the temperature range from average minimum to average maximum temperature. Data obtained from the Bureau of Meteorology, Combienbar weather station.

East Gippsland forests

All of Far East Gippsland falls within the two bioregions “East Gippsland Uplands” and “East Gippsland Lowlands”. The floristic composition of the forests varies markedly across these bioregions. Classification of the floristic communities into Ecological Vegetation Classes (EVC) was undertaken by the Victorian Department of Sustainability and Environment (Davies *et al.* 2002; DSE 2010b). The dominant vegetation community in Far East Gippsland is Lowland Forest (EVC 16) and Damp Forest (EVC 29). These are characterised by a eucalypt tree layer to 30 m tall consisting of silvertop ash (*Eucalyptus sieberi*), white stringybark (*E. globoidea*), messmate stringybark (*E. obliqua*) and mountain grey gum (*E. cypellocarpa*). The dense understory consists of layers of shorter trees and shrubs such as blackwood (*Acacia melanoxylon*), narrow-leaf geebung (*Persoonia linearis*) and tangled guinea-flower (*Hibbertia empetrifolia*) (DSE 2010a).

In the northern areas of Far East Gippsland, the soil composition changes to a well drained sandy soil in the east and shallow infertile soils on the north western highlands. This geological change is also reflected in the vegetation communities, which become banksia woodlands (EVC 14) in the north east dominated by saw banksia (*Banksia serrata*) to 10 m tall, with a thick understory. In the north western highlands the shallow infertile soils are typically characterised by open forest to 25 m tall with yellow stringybark (*E. muelleriana*) mountain grey gum (*E. cypellocarpa*) and red stringybark (*E. macrorrhycha*) dominating. The understory in these areas consists of a low shrub layer and tussock-forming graminoids such as thatch saw-sedge (*Gahnia radula*) and forest wire-grass (*Tetrarrhena juncea*) (DSE 2010a).

Industry

According to the 2006 census, the largest employer groups in the region are the retail trade industry (2 047 people), health care and social assistance (1 810 people) and agriculture, forestry and fishing (1 383 people) (ABS 2008). Three main industry groups - tourism, agriculture and timber - contribute $471.8 million towards the gross value output for East Gippsland (Cameron *et al.* 2005), or 61% of the regional total.

The timber industry in the region was relatively small until the 1950s when it became a major activity and employer and the mainstay of many of the smaller towns (NRE 2010). About one third of the public land in Far East Gippsland is available for timber harvesting with some 6 000 ha of forest harvested annually (DSE 2010a). On average, 3.4 million cubic metres of sawlogs and woodchips are harvested annually in the Gippsland region as a whole, with half of this is processed within the region. Gippsland sawmills process about 515 000 m^3 of softwood sawlogs into 221 000 m^3 of sawn timber annually. The region exports some 75 000 m^3 of softwood sawlogs and approximately 720 000 m^3 of woodchips annually (Cameron *et al.* 2005).

There are concerns that the timber industry is having negative impacts on the biodiversity within the region (DSE 2010a). Many of these impacts are likely to arise as a result of changes in the forest structure. As old growth trees are removed, the forest becomes dominated by young regrowth and the resultant homogeneous age structure reduces the vegetative complexity within the forest. Under the current forestry management regime, many regrowth trees will be harvested before becoming old enough to form hollows for birds and mammals (Lindenmayer *et al.* 1990; Lindenmayer *et al.* 1999).

Human population

East Gippsland was home to five separate indigenous language groups before European settlement. Prior to 1800, the highest densities of the indigenous people in the region were along the coastline and rivers but numbered only a few thousand (NRE 2010). European settlement of the region commenced in the late 1830s. Settlement and population growth in the region were slow until the mid 19th century but increased rapidly following the discovery of small gold deposits and the arrival of the railway from the west (NRE 2010). The 2006 census profile for "East Gippsland (S) – Orbost (Statistical Local Area)" (ABS 2008), which corresponds roughly with the region "Far East Gippsland", shows that there are currently 8 298 people living in the area with approximately 3.6% identifying themselves as indigenous.

Southern Ark project

Far East Gippsland is the site of two major programs that have sought broad-scale reductions in invasive predator abundance. The first, Project Deliverance, commenced in 1998 to determine whether on-going, low intensity baiting would reduce the abundance of the red fox (*Vulpes vulpes*) and stimulate a commensurate increase in the abundance of five indicator species (Murray *et al.* 2006). Two of those species, the long-nosed potoroo (*Potorous tridactylus*) and southern brown bandicoot (*Isoodon obesulus*) had significantly larger populations following fox management at the end of the program compared with the non-baited control sites. Based on the outcomes of Project Deliverance, the Southern Ark project was established with the aim of reducing fox predation to enable the recovery of a range of native wildlife species (Murray *et al.* 2005).

Southern Ark is a large-scale fox management program encompassing about 1 million hectares of public land in Far East Gippsland. It encompasses all public land east of the Snowy River, bounded to the north by the Victorian / New South Wales border and to the south by the coastline (Murray *et al.* 2005) (Figure 3). Private land is not included in the program. The Southern Ark operation is divided into six management sectors which are treated as separate units within the program. Six monitoring areas were established within these management sectors to monitor the effectiveness of the baiting campaign. Three were designated as poisoned monitoring areas (treatment sites). The remaining three were designated as un-poisoned monitoring areas (non-treatment sites) and act as experimental control sites.

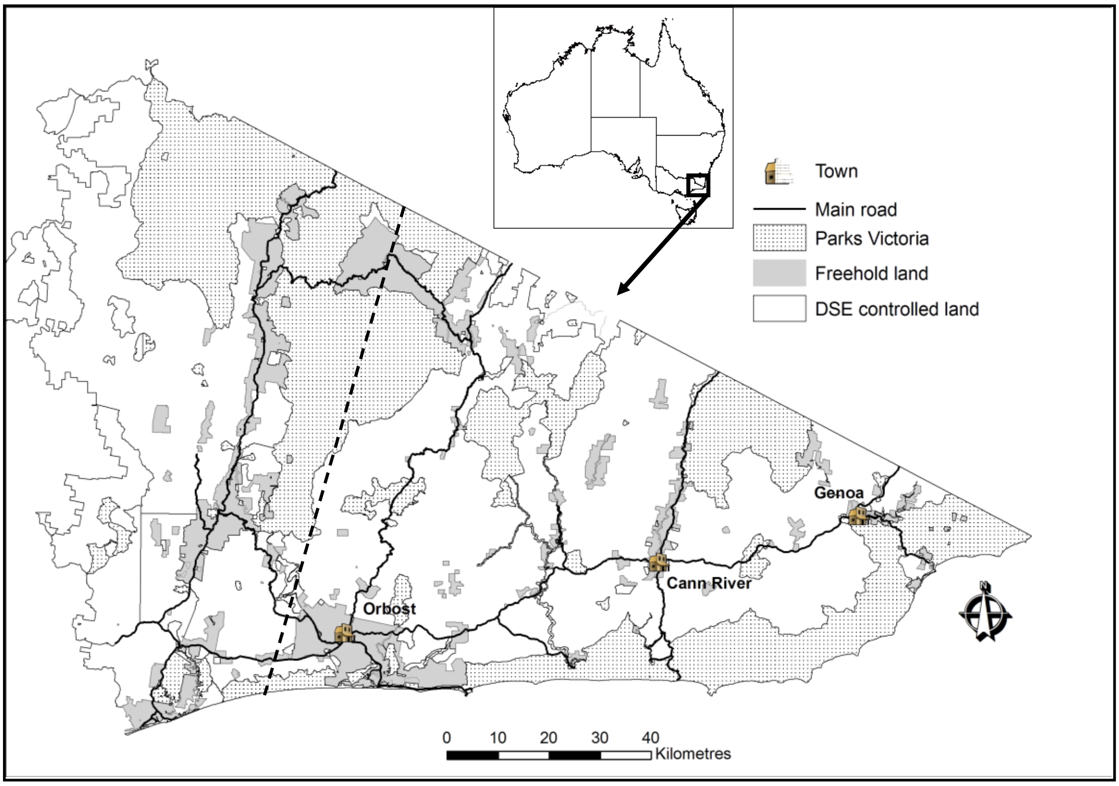


Figure 3. Map of the study area. The broken line indicates the approximate western boundary of the Southern Ark project. Background GIS data layers were provided by the Victorian Department of Sustainability and Environment.

Southern Ark has established 3 486 bait stations throughout the six management sectors at which poison baits are set to reduce fox numbers. Bait stations were constructed at 1 km intervals along forest roads. Each station consists of a 1 m circular depression filled with sifted loamy soil or coarse river sand. Foxoff® baits (Animal Control Technologies Australia, Somerton, Victoria) are buried 15 cm below the surface and a tuna oil emulsion is added as an olfactory lure (Murray *et al.* 2005). Baiting occurs only on accessible roads on public land within the area of operation (Figure 4).

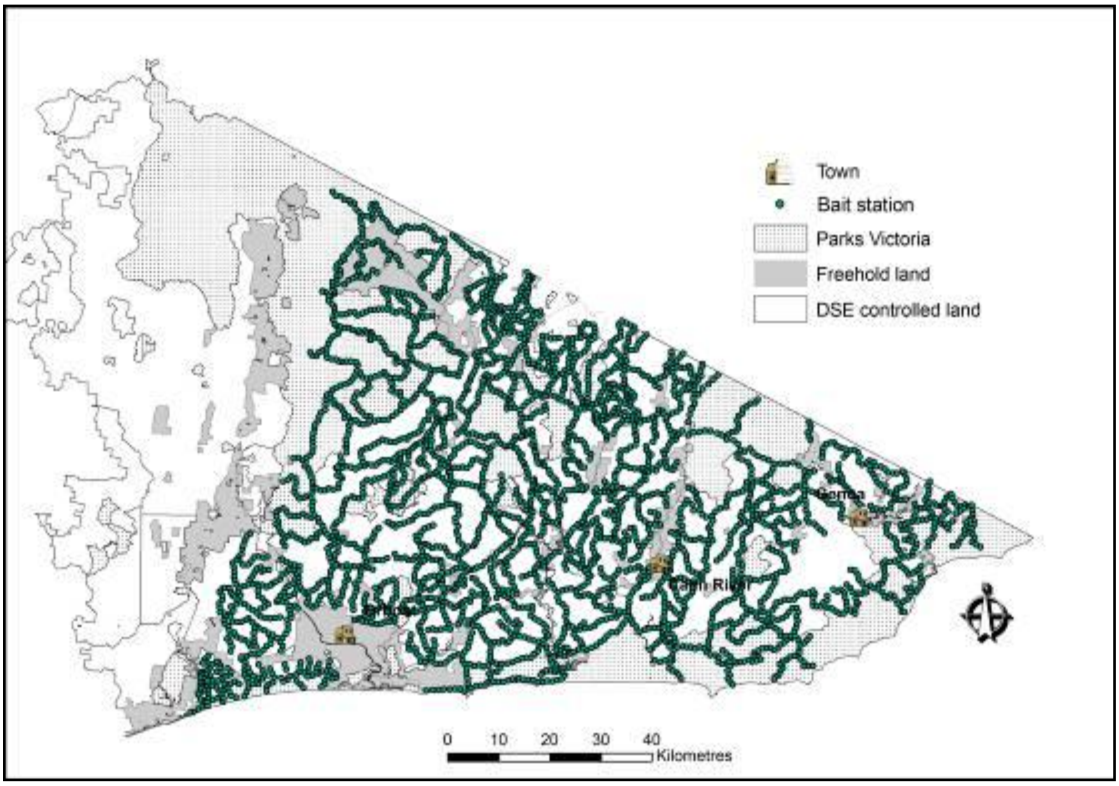


Figure 4. Location of fox-baiting stations in the Southern Ark area. GIS data sources from the Department of Sustainability and Environment.

Monitoring areas

The six monitoring sites were set up to monitor the responses of both foxes and selected prey species. The three sites designated as poisoned monitoring areas (PMAs) are baited in the same manner as for the rest of the Southern Ark Project area (Table 1). The remaining three non-treatment sites (un-poisoned monitoring areas or UMAs) have no poison baiting. To minimise potential bias, non-poisoned baits are laid in the UMAs in the same manner as the poisoned baits. The six sites were paired so each PMA had a corresponding UMA that would be monitored at the same time.

The Southern Ark project was reviewed extensively by Diment (2010), who also provided additional information on the efficacy of fox baiting and a preliminary assessment of the effects of fox reduction on prey species.

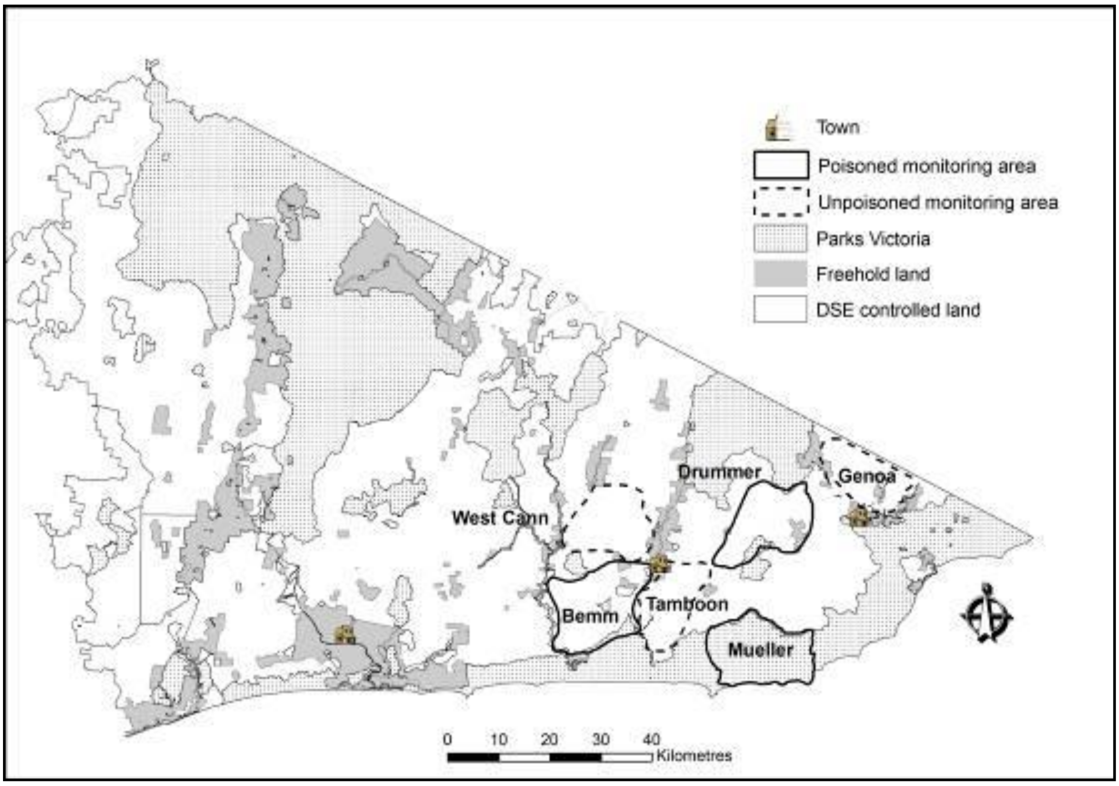


Figure 5. Monitoring areas in the Southern Ark project. The three un-poisoned monitoring areas are the only areas within Southern Ark that are not being baited, and act as non-treatment (control) sites.

Table 1. Monitoring areas within Southern Ark.

Poisoned monitoring area	Area (ha)
Mueller	23 100
Bemm	25 000
Drummer	20 500
Un-poisoned monitoring areas	
West Cann	19 000
Tamboon	23 100
Genoa	15 100

This study

The location of, treatment allocated to and pairing of the monitoring areas used in this study were decided by DSE management during the design phase of Southern Ark (Murray *et al.* 2005), and were established well before this project commenced. My study was conducted primarily in the West Cann UMA and the Drummer PMA

(Figure 5). Whilst not paired in the Southern Ark Project, these two monitoring areas were selected because they are geomorphologically more similar than other combinations of UMA and PMAs. Although these sites were chosen because of their geomorphological similarity, they differ in respect to fire history and logging regimes. No direct comparisons are made between these monitoring areas in this study and I have attempted to treat these areas as part of contiguous forest. Fox baiting did not occur in the Drummer PMA until late 2008. As all my data on the movement and home ranges of feral cats had been gathered by the time baiting commenced, I assume that no bias was introduced as a result of differing fox management treatments being allocated to these areas.

Fire

In 1983 a wildfire burnt large portions of Far East Gippsland. The Drummer PMA was completely burnt. More recently, a fire in 1995 re-burnt the north eastern quarter of this monitoring area. In comparison, only the north eastern quarter of the West Cann UMA was burnt in the 1983 fire. In 1990 a large fire burnt a further quarter of this monitoring area, in the south-east corner. There was very little overlap of these two fires. The only other fire event in this area occurred in the early 1970s (Figure 6).

Logging history

Over half the area available for harvesting in East Gippsland consists of trees over 150 years old with the balance made up of recent regeneration from harvesting since 1965 (Cameron *et al.* 2005). Logging is undertaken in coups of approximately 30-35 ha. Logging rotation age (the time between harvesting of the same coup) is 83 years for silvertop ash (*Eucalyptus sieberi*) and 112 years for coups of mixed species. Both West Cann UMA and Drummer PMA are logged using clear-felling techniques (Cameron *et al.* 2005; DSE 2010a). These areas have been logged at different rates since 1960 with nearly half of West Cann UMA having been logged in the past 50 years compared with about one quarter of the Drummer PMA (Table 2).

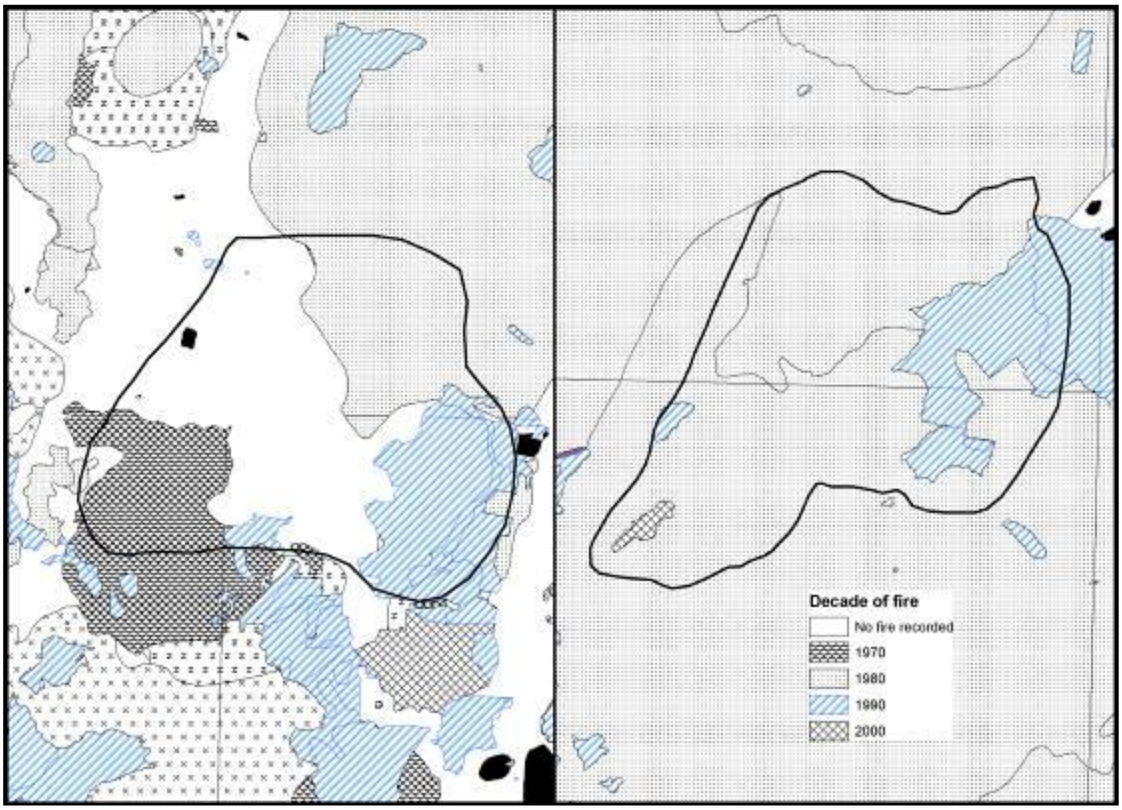


Figure 6. Fire history of the West Cann and Drummer monitoring areas shown by decade. West Cann UMA is shown on the left and Drummer PMA on the right. Data for this figure were extracted from GIS fire layers provided by the Victorian Department of Sustainability and Environment.

Table 2. Area logged within the West Cann and Drummer monitoring areas by decade. Data were extracted from GIS layers provided by the Department of Sustainability and Environment, Victoria

Decade of harvesting	West Cann UMA		Drummer PMA	
	ha	%	ha	%
Pre 1930	223	1.2	21	0.1
1960-69	3 462	18.2	21	0.1
1970-79	2 263	11.9	934	4.6
1980-89	636	3.3	2 185	10.7
1990-99	1 526	8.0	1 558	7.6
2000-10	1 037	5.5	489	2.4
Total	9 147	48.1	5 208	25.4

Vegetation communities

The two monitoring areas have differing floristic compositions based on Ecological Vegetation Classes (EVC) (Table 3). Lowland forest and damp forest are the dominant vegetation communities within the Drummer PMA while damp forest and dry shrubby forest are the dominant vegetation communities in the West Cann UMA (Davies *et al.* 2002). The forest in each area is dominated by messmate (*E. obliqua*), silvertop ash (*E. sieberi*), brown barrel (*E. fastigata*), and white stringybark (*E. globoidea*).

Table 3. Composition of West Cann and Drummer monitoring areas based on the four most common Ecological Vegetation Classes (EVC) within those areas. *Less common EVCs are not shown so totals do not equal 100%

Ecological Vegetation Class	West Cann UMA		Drummer PMA	
	ha	%	ha	%
Lowland forest	3 571	18.7	9 795	47.8
Shrubby dry forest	4 035	21.2	235	1.1
Damp forest	9 499	50.0	5 270	25.7
Wet forest	1 147	6.1	1 581	7.7
total	1269	96.0*	1092	82.3*

Aims and structure of this thesis

In this thesis I examine the ecology of feral cats in a tall forest ecosystem and then explore potential control techniques for feral cats in this and other habitats. I investigate the home range sizes and patterns of home range use by feral cats in the Southern Ark area. I also examine a technique based on poison baiting for managing feral cats and the potential for uptake of baits by non-target species using a decision tree process. My broad goal is to fill one of the gaps in the body of knowledge of feral cats; that is, to document aspects of their ecology in tall forests and to examine the potential for an innovative control technique that is applicable to a broad range of species and situations.

My specific objectives are to

I. Review the introduction, spread and impacts of feral cats in Australia (Chapter 2)

I discuss how the domestic cat came into Australia and how it spread across the landscape. I also review the potential impacts of feral and domestic cats both on the mainland and offshore islands. This section provides a synthesis of the current knowledge of feral cats in Australia.

II. Examine the ecology and movement of feral cats in tall forests (Chapters 3-4)

No previous research has been undertaken on the ecology of feral cats in tall closed forests. These chapters fill that knowledge gap to some extent through the use of GPS collars to obtain location data. GPS collars allow movement patterns and foraging behaviours to be examined in more detail than is usually possible with conventional VHF telemetry. I scrutinise patterns of intra-home range use by feral cats using a novel analysis technique and discuss the potential influence of cat space-use on prey species.

III. Examine how feral cats utilise their home ranges (Chapters 5)

Burt (1943) describes an animal's home range as the area "....traversed by the individual in its normal activities of food gathering, mating, and caring for young…". Very few animals utilise the whole of their home range. Here I examine the large unused areas within the home ranges of feral cats and propose possible explanations as to why these areas are not used.

IV. Examine the population demographics of feral cat prey species (Chapter 6)

Baiting programs designed to reduce feral cat abundance need to target times when cats are food stressed. During these times cats are more likely to scavenge for food and will consume poisoned baits. In south-eastern Australia, this generally occurs in winter when the natural fluctuations in prey abundance are at their lowest, particularly following the male die off in *Antechinus* species. In this chapter, I undertake small mammal trapping at four sites to determine what prey species are available to be consumed by feral cats and at what time their abundance is at a low ebb to determine when a poison baiting campaign for feral cats would be most effective.

V. Investigate a potential technique for managing feral cats (Chapters 7-8)

There are few techniques available for the management of feral cats, and none that is cost-effective. However, poisoning is being investigated currently as a potential means of primary cat control. Access to poison bait by non-target species can restrict the use of poisons and limit the probability of success of a management program. In these chapters I present a desktop analysis using a decision tree process to determine which non-target species are likely to encounter baits, and which are likely to access the toxicants that are encapsulated within a hard shell delivery vehicle (HSDV) inserted in the bait. I also report on field experiments using a non-toxic bait marker to compare the access that non-target species have to toxicants enclosed within a HSDV compared with that directly injected into the bait media.

VI. Integrate my findings into the current pool of knowledge of feral cats (Chapter 9)

In this chapter I present a synopsis of the key findings of my research and integrate the relevant findings into current knowledge of feral cats.

Chapter 2

The feral cat (*Felis catus*): a review of its history, introduction into Australia and ecological effects

Chapter overview

In this chapter, I review current knowledge of the history and ecology of feral cats (*Felis catus*) including information on the damage that they cause and trends in thinking regarding the management of this ubiquitous species. The review covers research on cats in different parts of the world but maintains an overall focus on cats in Australia as this is where my own primary research was carried out. I also include a small section on the possible impact of domestic cats on native wildlife in the urban environment.

The cat has had a long and varied relationship with humans over the past 10 000 years. It is thought that initially cats were used as a source of food and pelts before eventually being tamed and subsequently domesticated. The cat was revered in ancient Egypt to the point of becoming a deity and having a ban placed on its export. By the time of the Roman Empire, domestic cats were part of everyday life and had dispersed across Europe with the expansion of the imperial empire. The subsequent movement of people across the globe allowed the domestic cat to extend its range and establish permanent populations on all continents except Antarctica. Cats were introduced to Australia with European colonisation. Domestic cats were introduced both accidentally and intentionally into the wild and have subsequently spread across the mainland, to Tasmania and many offshore islands.

Even with a long history of domestication, feral cats still possess all the hunting instincts and abilities of their forebears which allow them to survive easily in a feral state. Feral cats can impact negatively on native wildlife in many ways, including via predation and as carriers of diseases and parasites. Feral cats have had significant negative impacts on attempts to reintroduce endangered species back into their natural ranges. Counter-intuitively though, feral cats potentially may have a positive impact on some species through the suppression of lower-order predators or herbivores such as rats (*Rattus* spp.).

Managing feral cats to protect endangered native species is difficult primarily due to the limited techniques that are available, especially in states where effective control techniques have been banned by legislators. However, on-going research into new bait media and novel ways of delivering toxins that minimise the access of non-target species to toxicants hold promise for the future.

The cat holds a unique place in human society. It is both a loved family pet and a despised invasive predator. While there is generally strong support within society to manage feral cats, there are other attitudes that can disrupt or impair attempts to manage cats, such as the misconceived belief that feral and domestic cats are separate species or through a misplaced affection for feral cats.

History of the domestic cat

There has been a complex and changing relationship between humans and felids over the preceding 10 000 years. Bone and tooth fragments from *Felis lybica* (African wildcat) have been found in the excavations of early Jericho, but there is insufficient evidence to determine if these were the remains of domesticated animals or wild animals killed for food or pelts (Serpell 2000). More than 38% of all felid species have been tamed by humans over the millennia for one purpose or another, yet only *Felis silvestris* (European wildcat and its subspecies) has been domesticated (Faure and Kitchener 2009). Genetic analysis of over 1 000 felid samples from both wildcat species and domestic cats has indicated that all modern domestic cats originated from the Middle Eastern wildcat - *F. s. lybica* (Driscoll *et al.* 2009). Domestication is a gradual rather than sudden process and cats may have been captured, or kittens found and tamed as pets prior to the establishment of agricultural settlements (Serpell 2000).

Following the shift towards agricultural practices by humans in the Fertile Crescent of Mesopotamia, some *F. silvestris* probably became attracted to settlements to feed on rodents exploiting stored food items. Those cats may have been encouraged to stay to protect the food stocks, which eventually led to full domestication (Driscoll *et al.*

2007; Faure and Kitchener 2009). A complete cat skeleton found in a human burial site in a Neolithic village in Cyprus has been dated to ~9 500 years before the present (ybp) (Vigne *et al.* 2004). The non-disarticulation of the bones indicates that the cat had been intentionally buried rather than consumed as a food resource. As there is no evidence of any native felid on Cyprus, this indicates that cats must have been introduced to the island, and lived in close association with the humans of that time (Vigne *et al.* 2004). The finding of the bones of a small *F. chaus* (Jungle cat) in a pre-dynastic burial site in Egypt dated to 5 700 ybp indicates that cats were being tamed at that time (Linseele *et al.* 2007; Linseele *et al.* 2008). The bones were not disarticulated and showed signs of healing fractures, thus indicating that the animal had been in captivity for a number of weeks.

Examination of the genome of the cat indicates that domestication may well have taken place on multiple occasions in separate locations (O'Brien *et al.* 2008). The earliest evidence for full domestication of the cat (*Felis catus*) comes from ancient Egypt. Approximately 2 900 ybp the cat became an official deity of Egypt in the form of the goddess Bastet. The large number of sacrificed and mummified cats found in tombs and burial sites indicates that cats were being bred by Egyptians at that time (Driscoll *et al.* 2009). Cats were so revered by Egyptians that bans were placed on their export to other countries. However, this ban was ineffectual as by 2 500 ybp, domestic cats had made their way to Greece. By 2 300 ybp cats are shown in paintings and frescoes living with and as part of human society. The domestic cat expanded across Europe as the Romans expanded their empire some 2 000 years ago (Driscoll *et al.* 2009).

The Romans were probably responsible for the introduction of the domestic cat into northern Europe, with domesticated cats present in Britain by the fourth century AD. By 1 450 years ago it is probable that cats were fully domesticated and that the presence of a domestic cat within a household was commonplace. Cats were widespread throughout most of Europe and Asia by the tenth century AD (Serpell 2000). Subsequent movements of humans and their associated domestic animals have spread the domestic cat across the globe. By the time of the industrial revolution, cat owners were selectively mating their pets to produce new breeds of domestic cat

(O'Brien *et al.* 2008). Despite the progress of the cat into a domesticated state, it has retained its pre-domestication repertoire of behavioural and hunting traits that make it very successful when it reverts into a feral state (Bradshaw *et al.* 1996).

Introduction of the cat into Australia

There are many hypotheses on when cats were introduced into Australia. These range from cats having arrived with the indigenous people as much as 50 000 ybp, through Macassan trepangers in the 1600s (Baldwin 1980), or as a result of shipwrecks / coastal landings of European sailors and traders (Burbidge *et al.* 1988). While it is possible that single or small groups of cats arrived at various times prior to European settlement, it is most likely that the domestic cat did not arrive in Australia in sufficient numbers to form breeding populations until after European settlement. The domestic cat was most likely introduced into Australia at multiple points along the coastline during the period 1824-1886 (Abbott 2002). Cats then diffused across the Australian mainland from their points of entry along the coast and rapidly moved inland with European colonisation of these areas. Many early explorers failed to note cats at anywhere other than near settlements, indicating that cats were not widespread across Australia during the early stages of European settlement (Abbott 2002).

The successful introduction of the European rabbit (*Oryctolagus cuniculus*) to Australia and its rapid growth into plague proportions is very likely to have unwittingly accelerated the spread of the cat across Australia. Not only did rabbits provide a readily available food source, domestic cats were advertised, sold and released in their hundreds as a measure to combat the rabbit plagues (Rolls 1969). Due to a combination of intentional and unintentional releases of domestic cats into the wild, feral cats are now widespread across the Australian mainland, Tasmania and many offshore islands (Abbott 2002; Burbidge *et al.* 1997).

Dietary requirements and predatory behaviour

The cat is an obligate carnivore (Bradshaw *et al.* 1996). It descended from specialised predators that utilise large canines to sever the neck vertebrae of mammalian prey (Baenninger 1978; Bradshaw 2006). Cats are fundamentally more constrained in diet choices than canids due to the lack of certain key metabolic enzymes (Zoran 2002). These enzymes appear to have been lost in a common ancestor of all species in the cat family. This loss has resulted in a very narrowly defined diet that is high in protein and moderate amounts of fat, but low in carbohydrates. In the wild, this can be satisfied only by a diet that consists largely of vertebrate prey (Bradshaw 2006; Zoran 2002).

Hilmer (2010) showed that the basal metabolic requirements of a 3.7 kg feral cat in winter were about 800 kJ per day. Based on calculations by Nagy *et al.* (1999) this equates to a minimum 160 g of wet food intake per day for a cat of that weight. Feral cats are solitary hunters and generally take prey with a much lower body mass than their own. This can necessitate several kills per day to fulfil metabolic requirements.

Felis catus is a generalist resident predator that exploits a wide range of prey and is able to readily switch from one prey to another (Fitzgerald and Turner 2000). Feral cats hunt using a number of audio and visual cues (Fitzgerald and Turner 2000) and have a hunting strategy that relies heavily on crypsis for success. Cats also use both "mobile" and "stationary" hunting strategies to secure prey. A mobile hunting strategy involves actively seeking out prey items by moving through the environment. When a potential prey item is detected either through visual or audio cues, the cat starts to stalk the animal, exploiting whatever cover is available (Fitzgerald and Turner 2000). Jones (1977) found that cats on sub-Antarctic Macquarie Island used a mobile hunting strategy when searching for rabbits. Cats would move between rabbit burrows, entering and searching each one prior to moving on to the next. In contrast, a stationary (sometimes called "sit and wait") hunting strategy involves moving to an area that prey frequent, such as burrow entrances or animal trails, and laying in wait for a target to appear. When prey appears, it is ambushed from cover. Cats have been known to utilise this strategy when hunting for turtle hatchlings (Seabrook 1989), rabbits and burrow-nesting seabirds (Corbett 1979 as cited in Fitzgerald and Turner (2000)).

Potential impacts caused by cats

Feral cats

The predominant damage attributed to feral cats is through predation on birds, mammals, reptiles and amphibians, although some probably also occurs through competition for available food and shelter resources (Burbidge and Manly 2002; Dickman 1996). A worldwide meta-analysis which included multiple predator species, showed that predator management had positive effects on populations of non-cyclic prey species (Salo *et al.* 2010). The effect of predator management was variable for cyclic prey species (those that go through boom and bust phases). Predator management resulted in a positive effect on prey species populations when those prey were at low densities but had no obvious effect when prey populations were at their peak. The negative impacts of introduced predators on native prey populations are usually greater than those of native predators (Mcevoy *et al.* 2008; Salo *et al.* 2007). While many marsupial predators did, and currently do, exist in Australia, placental carnivores were not present in Australia until introduced by humans 3 500 – 5 000 years ago (Johnson and Wroe 2003). As a result, Australian marsupials did not co-evolve with eutherian predators. Prey species that co-evolve with the predator possess predator recognition and avoidance strategies that naïve prey species lack (Banks 1998; Mcevoy *et al.* 2008; Russell and Banks 2007). The lack of strategies for predator avoidance can lead to introduced predators having substantial negative impacts on populations of naïve native prey species.

Feral cats have been implicated in the decline and extinction of a number of species of native Australian mammals (Burbidge and Manly 2002; Dufty 1994; Smith and Quin 1996). There is also evidence of the deleterious impacts of feral cats on Australian islands that has been obtained through modelling island species populations (Burbidge and Manly 2002; Burbidge *et al.* 1997). Feral cats are listed as a threat to 117 native species in New South Wales (Coutts-Smith *et al.* 2007). Smith and Quin (1996) suggested that the presence of feral cats is the best predictor of the decline in abundance of conilurine rodents.

There is a positive correlation between the presence of feral cats on Australian islands and the extinction of ground-dwelling native fauna. This relationship is particularly strong on arid islands that lack refugia, such as rock piles, for the native animals (Burbidge and Manly 2002). Dickman *et al.* (1993) found that predation by feral cats could be implicated in the regional extinction of up to 10 species of native small mammals in western New South Wales prior to 1857. Predation by feral cats together with habitat degradation through overgrazing by stock was suggested by Smith *et al.* (1994) as instrumental in the extinction of the black bittern (*Ixobrychus flavicollis*) and Lewin’s rail (*Dryolimnas pectoralis*). Smith *et al.* (1994) also concluded that the primary cause of decline of avifauna in the New South Wales western division was habitat degradation, with predation by exotic animals as a secondary cause. The spread of an epizootic disease across Western Australia has been suggested as a primary cause of the collapse of native fauna populations in this region, with feral cat predation and drought being additional influencing factors (Abbott 2006).

Few studies have examined the actual impacts of feral cats on prey species at a population level. At Heirisson Prong, Western Australia, Risbey *et al.* (2000) showed that when fox density was reduced, the feral cat population increased and the small mammal population declined by 80%. This remains the only study in Australia to show the impact of feral cats on established native species populations through experimentation rather than by analysis of historical records or recovery of populations post-feral cat reduction.

Feral cats affect prey species populations most directly through predation. Multiple dietary studies have shown that cats prey on native species (for example - Brunner *et al.* 1975; Coman and Brunner 1972; Glen *et al.* 2010; Jones and Coman 1981). However, inclusion in the diet does not necessarily mean that cat predation is having a detrimental impact at the population level that in turn leads to a decline of the prey species (Denny and Dickman 2010). An impact will occur only if the level of harvest by the predator exceeds the rate of increase of the prey species (Krebs 2001). For example, Banks (1999) found that fox removal had no effect on bush rats (*Rattus fuscipes*) and surmised that the foxes were taking the “doomed surplus”, i.e. that portion of the population that would have perished anyway even in the absence of foxes. Care must be taken not to assume that feral cats are having a negative impact on prey populations simply on the basis of the inclusion of these prey in the diet. While recovery of a population post-cat reduction may indicate that the cats were controlling the population at that level, it does not necessarily mean that they were the cause of the initial decline as other correlated or unmeasured factors could well have been involved (Hone 1999a).

Some studies on islands show cats as the definitive cause of extinction of prey species. A single cat was responsible for the extirpation of the Angel de la Guarda deer mouse (*Peromyscus guardia*) on Estanque Island in the Angel de la Guarda Archipelago. A previously unrecorded population of *P. guardia* was discovered on Estanque Island in October 1995 and described as “relatively abundant” (Vazquez-Dominquez *et al.* 2004). By November 1998, the population had become extinct (Mellink *et al.* 2002). A single cat was found and removed from the island. Analysis of cat scats collected from the island found that 93% contained *P. guardia* hair (Vazquez-Dominquez *et al.* 2004). Similarly, a single pregnant cat was introduced to Stephens Island, New Zealand, by the lighthouse keeper. This cat and her subsequent offspring are believed to be responsible for the extinction of the Stephens Island wren (*Xenicus lyalli*) (Galbreath and Brown 2004).

While there is much circumstantial evidence on the detrimental impacts of feral cats, it comes predominantly from population recoveries of prey species following cat management and from modelling of predation-based scenarios. There is little direct evidence of negative impacts by feral cats on prey species at a population level. Caution must be applied when interpreting prey species population increases following feral cat management programs. It is possible that some other factor(s) induced the decline (Hone 1994; Hone 1999a).

Environmental systems are complex, with many biotic and abiotic interactions occurring. Many factors other than predation, such as changed fire regimes, land clearing, grazing by domestic stock, drought and floods can impact on the persistence of native fauna. Our knowledge of these systems, and the interactions that occur within the systems, is very limited (Braysher 1993). It becomes easy to blame the introduced predator that includes the prey species in its diet for any population decline in the prey species (Olsen 1998). For example, fox predation was blamed for the decline in malleefowl (*Leipoa ocellata*) populations in central western New South Wales, but populations did not increase following fox management. Subsequently it was found that chicks were not surviving due to a lack of available food (Priddel and Wheeler 1990). Similarly, Hone (1999a) analysed rock wallaby (*Petrogale lateralis*) recovery following fox management (see Kinnear *et al.* 1988; Kinnear *et al.* 1998). The rate of population increase was lower than would be expected if fox predation was the sole limiting factor, thus suggesting that some other factor(s) was still limiting population increase.

Feral cats and reintroduction attempts

Feral cats have impacted negatively on attempts to reintroduce native fauna to areas where local extinctions have occurred. Soderquist (1995) found that feral cats killed between 39 and 50% of reintroduced brush-tailed phascogales (*Phascogale tapoatafa*) within the first week after release on two separate occasions in Gippsland, Victoria. The release of rufous hare-wallaby or mala (*Lagorchestes hirsutus*) into the Tanami Desert was compromised by a single cat at each of two release locations (Gibson *et al.* 1995; Gibson *et al.* 1994). Attempts to reintroduce burrowing bettongs (*Bettongia lesueur*) into the Gibson Desert similarly met with failure as feral cats killed the released animals. Within 60 days of release, no living bettongs could be found and all the remains that could be located showed clear signs of having been killed by cats (Christensen and Burrows 1995). In an early review of macropod reintroductions in Australia, Short *et al.* (1992) found that the reduction or exclusion of introduced predators was critical to the success of any reintroduction program and that feral cats were responsible for the failure of a number of attempted reintroductions. A more recent review has confirmed the role of predators, including cats, in causing the failure of many reintroduction programs, with predator absence being a key predictor of reintroduction success (Finlayson *et al.* 2010).

Most of the animals used in these reintroductions were reared in captivity (Gibson *et al.* 1994; Soderquist 1995) or translocated from predator-free islands (Christensen and

Burrows 1995). These animals had no previous exposure to predators. Training naïve animals to avoid predators is an essential step to improve the success of reintroduction programs, with many failing when this step is not undertaken (Kleiman 1989). McLean *et al.* (1995) instigated a predator aversion training program for captive rufous hare-wallabies following the quantification of the impacts that predators had on released captive-bred animals. However, it is unknown if this training was successful. A successful predator aversion training program was undertaken by van Heezik *et al.* (1999) which resulted in greater survival of captive-reared houbara bustards (*Chlamydotis macqueenii*) when compared with those which did not receive aversion training.

Amensal effects

Amensal effects occur where interactions between individuals result in the decrease in fitness of one but have no effect on the other. In the cases considered here, it is the cat that would suffer no decline in fitness. Feral cats can introduce and spread pathogens, such as the toxoplasma parasite (*Toxoplasma gondii*), and diseases through other species (Eberhardt *et al.* 2006; Henderson 2009; Hill and Dubey 2002). Dickman (1996) reviewed a number of parasites and diseases that could be transmitted from cats to native wildlife and concluded that two pathogens are of considerable concern – *Spirometra erinacei* and *Toxoplasma gondii*.

Spirometra erinacei is a large tapeworm that infests the gut of carnivores and uses cats as a definitive host. Eggs passed by the carnivore develop into procercoids in freshwater copepods and small crustaceans. If these are consumed they develop into the second intermediate stage, plerocercoids. These have been recorded in native mammals, reptiles and amphibians (Moodie 1995; Munday *et al.* 1978). While producing no obvious clinical symptoms, they can cause muscle and soft tissue damage which may lead to death.

Toxoplasma gondii also uses cats as a definitive host and causes toxoplasmosis. It is common among both the domestic and feral cat populations in Australia (Adams *et al.* 2008; Hartley and Munday 1974; Smith and Munday 1965). *Toxoplasma gondii* is usually transferred to herbivores that eat grass containing oocysts that were excreted as the cat defecated (Meireles *et al.* 2004). Insects such as flies and cockroaches can also carry the oocysts on their feet and mouthparts. Ingestion of oocysts can occur through consuming the insect or something it has walked over (Hartley and Munday 1974). Infection by *T. gondii* can result in the aborting of foetuses in pregnant animals (Hartley and Munday 1974; Plant *et al.* 1974) or cellular damage to the brain, heart, lungs and/or liver (Jensen *et al.* 1985). It has been found in many native animals in eastern and central Australia, including dasyurids, bandicoots, macropods and some birds (Johnson *et al.* 1988; Moodie 1995; Smith and Munday 1965).

The extent of the impact that pathogens and diseases carried by feral cats have on native species populations remains unclear. While they are present within both cats and native wildlife populations, their presence does not reveal what level of impact, if any, they have. Further research is needed into the effects that these diseases and pathogens actually have at the population level on native species.

The presence of predators can affect not just the population size of prey species, but also the behaviour of prey individuals (Charnov *et al.* 1976). Such behavioural shifts results in the prey animals making shorter and less frequent feeding forays from a place of shelter or using more protective microhabitats (Brown 1988; Kotler 1984a; Lima and Bednekoff 1999). When the risk of predation is lower, animals forage more widely, for longer periods and are able to obtain more resources (Brown *et al.* 1988; Kotler 1984b; Lima 2002; Lima and Bednekoff 1999). The continued presence of a predator, or predators, can result in prey animals suffering depressed foraging and hence declines in individual fitness (Arthur *et al.* 2004; Mitchell and Lima 2002; Ruxton and Lima 1997). Using a modelled scenario, Beckerman *et al.* (2007) predicted that increasing densities of cats in urban areas had a negative effect on the fecundity of song birds due to sub-lethal effects rather than through direct predation on the songbirds. However, this model included only cat density as an influence on bird fecundity and did not include any other effects of urbanisation that may impact on fecundity such as habitat simplification (van der Ree and McCarthy 2005) or the increasing habitat fragmentation that occurs towards the central business district in most urbanised landscapes (Collins *et al.* 2002).

Domestic cats

Whilst cats have been domesticated for many centuries, they have retained their repertoire of behavioural and hunting traits (Bradshaw *et al.* 1996). Domestic cats impact on fauna within their local ranges in a similar manner to feral cats. Predation is the primary impact of domestic cats. Soulé *et al.* (1988) described domestic cats as subsidised predators because their primary source of nutritional intake is not through predation. As domestic cats obtain all their food requirements from their owners, predation becomes a form of play rather than a necessity (George 1974). This potentially results in a greater impact on prey species than would occur if the predation was by feral cats. Classical optimal foraging theory suggests that when the cost of obtaining food is greater than the benefit gained, the predator will move to another patch to search (Charnov 1976; Pyke 1984). Being nutritionally subsidised, domestic cats can continue to hunt and catch animals in an area long after the prey population has been reduced to a level that cannot support feral cats which rely on wildlife for all their food requirements.

Barratt (1997) surveyed local cat owners in Canberra and found that 1 961 prey items were brought home by the 214 domestic cats in his study over a 12 month period. The primary prey items brought home were small mammals. The estimated amount of prey caught annually by domestic cats in Canberra was between 380 000 and 630 000 animals (Barratt 1998). The majority of these were introduced species with only a small proportion of native species. The predation rates of cats living closer to grasslands or rural settings were higher than of those in the centre of suburbs. In Felmersham, England, all domestic cat owners, except one, participated in a study for a period of 12 months to determine what prey items were returned by their cats. The 70 cats in the village brought home a total of 1090 prey items in the 12 month period (Churcher and Lawton 1987). As found by Barratt (1997), most prey items were small mammals.

Predation by domestic cats in Dunedin, New Zealand, has been suggested to impose an unsustainable harvest on the urban bird populations (van Heezik *et al.* 2010). Estimates of the total catch by domestic cats of six species of bird were either higher than the total, or close to the lower 95% confidence interval, of the urban population estimates for those species (van Heezik *et al.* 2010). Conversely, Grayson *et al*. (2007) found that predation by domestic cats in Perth was not a significant factor in the decline of passerine birds in the urban area, but rather the density of housing, distance to woodland and the size of the bushland remnants were more important factors in the declines. Cat predation following the control of coyotes in San Diego County, California was found to have contributed to the local extinction of many chaparral-dependent bird species that had persisted in remnant patches of chaparral (Soulé *et al.* 1988). Body condition, diet and the regularity of feeding do not appear to influence the reported frequency of predation by domestic cats (Barratt 1998; Robertson 1998).

While predation does occur by domestic cats, the impact of this on populations of prey species is largely unknown. For there to be a detrimental impact, predation by domestic cats needs to exceed the rate at which the prey population can replenish itself. If predation does not exceed the ability of the population to replace itself yet it is still declining then there must be other causal factors involved. It is unlikely that predation by domestic cats would be the sole cause of population declines in urban areas, except where endangered prey species are restricted to small remnant patches of suitable habitat, as was found to be the case by Soulé *et al.* (1988). The process of urbanisation results in the removal or altering of many obligate resources for native species (Collins *et al.* 2002; McKinney 2002; Olden *et al.* 2006). This habitat modification alone has been responsible for the reduction in native plant species, invertebrates, reptiles, birds and mammals in urban areas (for examples see Jellinek *et al.* 2004; Odell and Knight 2001; Rickman and Connor 2003; Williams *et al.* 2005). The decline of native small mammals in the urban reserves of Canberra was suggested to have occurred through a number of processes, including habitat modification and changed fire regimes, rather than just predation by exotic species (Buckmaster *et al.* 2010).

Selective hunting and prey specialisation

Past studies on the diet of feral or domestic cats have rarely examined the level of prey availability. As a result, it is not known if cats selectively prey on some species in preference to another or if they are opportunistic hunters that take prey in similar proportions to those available. Selective predation by foxes on broad-toothed rats (*Mastacomys fuscus*) was found to occur in Australian alpine areas (Green 2002; Green and Osborne 1981). Of the few studies that have examined prey availability, Molsher (1999) found that feral cats at Burrendong, NSW selectively preyed on rabbits (*Oryctolagus cuniculus*). They continued to prey preferentially on rabbits even after a 90% decline in the rabbit population following the arrival of rabbit calicivirus disease (RCD). Domestic cats in Canberra have been shown to selectively prey on house sparrows (*Passer domesticus*) and blackbirds (*Turdus merula*) over a range of other common bird species including starlings (*Sturnus vulgaris*) (Barratt 1997). Cats have also been shown to selectively depredate certain weight ranges within a prey population. Stray cats in Baltimore, Maryland, for example, preferentially took juvenile or sub-adult rats rather than the much larger adult rats (Childs 1986). Cats are also believed to prey at a greater rate on juvenile and sub-adult eastern barred bandicoots (*Perameles gunnii*)with low predation on adult animals in an urban setting at Hamilton in Victoria (Dufty 1994)

Prey specialisation occurs when an individual cat is more proficient at hunting and has greater success in capturing certain prey species over others (Dickman 2009). Some individual cats on Rottnest Island, Western Australia, were observed to have greater hunting success with mice, while others were more successful at hunting lizards or birds (Dickman 2009). Similar hunting specialisation was observed in a feral cat on North Head, Sydney that had a greater success rate when hunting rabbits compared with other prey species (Dickman 2009). Prey specialisation has been shown to occur in other predatory species such as African wild dogs (*Lycaon pictus*) (Kruger *et al.* 1999).

Further research is warranted into selective hunting and prey specialisation in feral cats. Gibson *et al*. (1995) surmised that a single cat killed most of the reintroduced rufous hare wallabies at each of the release locations in the Tanami Desert. Similarly, two boars accounted for most of the lamb predation events witnessed by Pavlov and

Hone (1982). In situations like these, the predation events will cease only when either the animals responsible are removed or when there are no prey animals left. Unless these particular animals are removed by management actions, no amount of population reduction will result in a decline in the impacts of predation.

Mesopredator release

Mesopredator release occurs when the management or removal of a high order or apex predator facilitates increased predation by a lower order predator, or mesopredator; this, in turn can result in increased impacts on shared prey species (Soulé *et al.* 1988). The so-called ‘release’ of the mesopredator can occur via increased activity or ranging behaviour in the short term, and by increased numbers if management of the apex predator is maintained over a long period. In Australia this interaction is most commonly thought to occur between the introduced red fox (*Vulpes vulpes*), the feral cat and the dingo (*Canis lupus dingo*). Reducing fox density is thought to release the cat from competition with, or control by, foxes, resulting in a greater impact on common prey species. Reducing fox abundance resulted in an 80% decrease in small mammals in plots where feral cats were not managed on Heirisson Prong, Western Australia (Risbey *et al.* 2000). Molsher (1999) also found that after fox reduction, feral cats were more active and used areas that were not used prior to the reduction. Reducing the densities of dingoes in the arid zone may well release the restrictions that dingoes impose on the foraging of foxes and feral cats, in turn resulting in greater impacts on a wide range of prey species (Johnson *et al.* 2007; Letnic *et al.* 2009).

Cats protecting birds

The effect of mesopredator release also may result in the counter-intuitive situation where feral cats actually protect endangered species. Rats (*Rattus* spp.) depredate the eggs of many bird species (Ettel *et al.* 1998; Grant *et al.* 1981). Feral cats utilise rats as a food source and are thought to limit the rat populations. Courchamp *et al.* (1999) suggested that reducing the abundance of feral cats may result in an increase in the level of rat predation on bird eggs and therefore result in an increase in damage to bird populations over that caused by the cats. The interactions between birds, rats and cats were further modelled by Fan *et al.*(2005) who found that two types of interaction can occur when cat densities are reduced. Firstly, severe mesopredator release can occur if cat removal results in a sharp increase in rat abundance and in turn leads to the extinction of the bird population. The second interaction is a milder mesopredator release effect that results in a negative impact on the bird population that does not lead to extinction (Fan *et al.* 2005). Similar negative impacts on shared prey species were found by Gambino *et al.* (2007) using a spatial modelling approach.

The negative impacts arising from managing feral cats have been disputed by Dumont *et al.* (2010). They modelled the effects of cats on long-lived seabirds and indicated that no scenario would prevent the extinction of Barau's petrel (*Pterodroma baraui*) from Reunion Island, Indian Ocean, unless the feral cat population was drastically reduced or eradicated. This was despite the presence of rats, a lower order predator, on the island.

There have been few manipulation experiments undertaken to determine if the "cats protecting birds" hypothesis has validity or not. In one such study, eradication of feral cats from Little Barrier Island, New Zealand, resulted in a decline in breeding success of Cook's petrel (*Pterodroma cookii*) due to increased rat predation on eggs. However, this effect was spatially heterogeneous and varied with altitude. It was not until the eradication of the Pacific rat (*R. exulans*) some 14 years later that petrel breeding success increased (Rayner *et al.* 2007). In Sydney, Dickman (2009) reported a strong negative relationship between predation on bird nests in trees (by rats, antechinus and ringtail possums) and the activity of cats. The eradication of feral cats from sub-Antarctic Macquarie Island resulted in substantial increases in the rabbit population despite on-going control efforts. The increase in rabbits in turn had significant detrimental impacts on native vegetation communities across the entire island (Bergstrom *et al.* 2009; Scott and Kirkpatrick 2008). Conversely, the eradication of feral cats from Ascension Island resulted in an increase in the sooty tern (*Onychoprion fuscata*) population despite the presence of rats on the island. No decrease in incubation success due to rat predation was detected (Hughes *et al.* 2008).

It is most likely that any mesopredator release effect involving cats, rats and birds is far more complex than a simplistic three-tier system might suggest and many additional factors will influence the outcome of any response by rats to cat control. It is likely that this issue will remain contentious until further experimentation is undertaken; even then it is unlikely that unambiguous results will be obtained that are applicable in all circumstances. Regardless, such uncertainty should not be used as a reason to refrain from managing feral cats to protect endangered species but should be taken into account when designing control programs. In particular, all potentially relevant second- and third-order interactions in the system should be identified and appropriate monitoring protocols should be included in any management plan.

Management of feral cats in Australia

Four techniques are commonly used to manage feral cats in Australia, and a fifth – trap-neuter-release – in other jurisdictions. On a per-project basis, the most widely used technique is trapping, followed by ground-shooting then poison baiting and exclusion fencing (Reddiex *et al.* 2006). In terms of areal coverage across Australia, feral cats are managed over more than 4000 km^2 annually, with most of this activity occurring through aerial baiting programs in Western Australia (Algar and Burrows 2004; Forsyth *et al.* 2005). While more cat management projects are undertaken in the eastern Australian states, they are on a much smaller scale than those in the Western Shield projects being undertaken in Western Australia (see, for example Algar and Burrows 2004; Morris *et al.* 2004), in the Arid Recovery project in South Australia (Moseby *et al.* 2009a), or those conducted by the Australian Wildlife Conservancy.

Techniques for controlling cats

Trapping: Trapping is the most commonly used technique for managing feral cats (Reddiex *et al.* 2006). Feral cats are trapped using both cage-style traps and rubber-jaw leg-hold traps. Steel jaw traps are banned in all Australian jurisdictions for feral cats due to welfare and ethical considerations. The far more humane rubber-jaw traps are just as effective at restraining captured animals, but do so without causing injury and also have fewer impacts on non-target species (Sharp and Saunders 2004c). However, amendments to Victorian State legislation in 2009 precluded the use of leg-hold traps of any nature for the capture of feral cats on public land. This has restricted the ability of cat management programs in Victoria to effectively reduce feral cat abundance. Despite this, leg-holds are still permitted to be used for the capture of rabbits, foxes and wild dogs.

Cage traps are relatively ineffective compared with leg-hold traps for capturing feral cats away from urban areas and are useful mainly in areas of high human disturbance such as rubbish tips (Sharp and Saunders 2004b; Short *et al.* 1995). Cats are naturally wary animals and may not enter an unfamiliar confined space such as a cage trap as it is novel and too different from their natural surroundings. However, Molsher (2001) found no difference in the use of cage traps and leg-hold traps when trapping feral cats at Burrendong in central New South Wales. The effectiveness of cage traps for feral cats is, at best, variable in areas with habitats that have not been highly modified by humans.

Shooting: Shooting is generally undertaken at night using a competent, qualified shooter with the aid of a spotlight. Shooting is a humane way of controlling feral cats (Sharp and Saunders 2004a), but it is both labour and time expensive and is ineffective in areas with thick undergrowth. Also, many programs that use shooting as a management technique do not do so in a strategic or systematic manner but rather in an opportunistic or *ad hoc* manner that further limits the effectiveness of the program (M. Braysher, pers. comm.).

Poison baiting: Poison baiting is used to reduce the numbers of feral cats over large areas, primarily in Western Australia. Generally poisoned baits are dropped from an aircraft flying on a predetermined path at a specified altitude (Algar and Burrows 2004; Burrows *et al.* 2003). At present the toxicant used for poison baiting is 1080 (sodium monofluoroacetate) due to its high toxicity to introduced eutherian mammals and the evolutionary tolerance of many native animals to the substance in the north and west of Australia (McIlroy 1981b; Twigg and King 1989). The tolerance of many Australian native animals arises from the presence of fluoroacetate compounds in 35 widespread species of native plant species. Thirty-three of these belong to the genera *Gastrolobium* or *Nemcia* with two additional species from the genus *Acacia* (Twigg and King 1991; Twigg *et al.* 1996; Wright *et al.* 1999). This tolerance allows baits to be laid on the ground surface in areas where these plants occur without the risk of significant adverse affects on native animal populations (Calver *et al.* 1989a; Eason and Frampton 1991; McIlroy 1981a).

This tolerance to 1080 does not extend to animals in the south and east of Australia as these species of plants are either not present or are sparse in those areas. Baits containing 1080 must be buried to prevent access to the toxicant by non-target species (DEWHA 2008). Burying baits does not deter canid species as these are able to detect buried baits using their well-developed olfactory senses (Fleming *et al.* 2001; Saunders *et al.* 1995). Cats do not possess the same olfactory acuity as canids and are less able to detect the buried baits. Additionally, as cats will not exhume buried baits, this method is not an effective way of reducing feral cat abundance (Seebeck and Clunie 1997).

Despite its widespread use, there is some perception that the use of 1080 is inhumane due to the clinical symptoms that can occur in the early stages of toxicosis, such a trembling, retching and uncontrolled paddling of limbs (Marks *et al.* 2000, Twigg and Parker 2010). This has resulted in increasing public aversion to the use of 1080 in poison baiting programs (Fitzgerald *et al.* 2007; Marks *et al.* 2000). A synthetic toxicant, para-aminopropiophenone (PAPP, see Savarie *et al.* 1983) is being trialled for use with feral cats (Johnston *et al.* 2011; Johnston *et al.* 2010b; Murphy *et al.* 2007). PAPP is also being trialled for use with other introduced predators such as foxes (Marks *et al.* 2004), stoats (*Mustela erminea*) (Eason *et al.* 2010b) and wild dogs (*Canis lupus*) (Murphy *et al.* 2007). PAPP is a methaemoglobin-forming compound that rapidly restricts the oxygen-carrying capacity of haemoglobin in the blood (Vandenbelt *et al.* 1944). At sufficient concentrations it produces anoxia (Eason *et al.* 2010b; Marks *et al.* 2004). Unlike 1080, an antidote is available to reverse the effects of PAPP toxicosis. Susceptibility to PAPP toxicosis varies widely between genera (Murphy *et al.* 2005; Savarie *et al.* 1983). The susceptibility of non-target species to PAPP toxicosis is currently being tested in both pen and field trials (Eason *et al.* 2010a; Fisher *et al.* 2008; Murphy *et al.* 2007)

Feral cats, other than those at garbage tips, rarely scavenge for food and prefer live prey (Algar *et al.* 2007; Leyhausen 1979). As a result, they will usually consume baits only when they are food-stressed and have to resort to scavenging to survive. These periods are most likely to occur during winter when prey abundance is low or following a boom period in prey availability in arid areas when prey populations are declining (Algar *et al.* 2007; Letnic and Dickman 2010).

Baits used for feral cat management need to be attractive and palatable to the cats to maximise the likelihood that they will consume them (Bradshaw *et al.* 1996). Dried meat baits such as those used for managing foxes and wild dogs are rarely consumed by cats (Risbey *et al.* 1997). Experiments with day-old chicks, fishmeal baits, chicken meat sausages and rabbits killed with 1080 to effect secondary poisoning have had limited or no success (Algar *et al.* 2007; Risbey *et al.* 1997). The use of frozen laboratory mice with a single grain of 1080-laced oats place at the back of the throat proved effective in reducing cat numbers (Short *et al.* 1997). However, preparing sufficient mice for broad scale aerial baiting would be both costly and time consuming.

The Western Australian Department of Environment and Conservation has recently developed the Eradicat® feral cat bait (Patent No. Au781829) (Algar and Burrows 2004; Burrows *et al.* 2003; Hetherington *et al.* 2007). The bait substrate is a soft, meat-based sausage (much like a chipolata) containing kangaroo meat, chicken fat and digests, and weighs approximately 15 g. Baits are automatically injected with 4.5 mg of 1080 during production and then air dried and stored frozen until used. The baits are attractive and palatable to feral cats both in Western Australia and in south-eastern Australia (Algar and Burrows 2004; Johnston *et al.* 2007). The Eradicat® bait is used extensively in Western Australia for managing feral cats over large areas through aerial delivery.

The Eradicat® bait is not target-specific and is consumed by a wide range of species including varanids, corvids and many native mammals (Burrows *et al.* 2003: Algar *et al.* unpublished data; also see Chapter 7 of this thesis). To minimise bait take by such non-target species, trials are underway to encapsulate the toxicant within a hard shelled delivery vehicle (HSDV) which is subsequently inserted directly into the bait media (Hetherington *et al.* 2007; Johnston *et al.* 2011; Marks *et al.* 2006). The HSDV is a proprietary product made by Scientec Research Pty Ltd (Melbourne Victoria) and is specifically designed to encapsulate toxicants (Provisional Patent No. 200890357). The HSDV consists of a hard, acid soluble polymer about 6 mm in diameter. The polymer is designed to dissolve quickly in stomach acid, thus rapidly releasing the toxicant into the gut following ingestion (Johnston *et al.* 2010b).

Feral cats lack grinding premolars and as a result, do not have a dentition that is suitable for chewing (Jones 1989). This results in cats swallowing larger portions of soft food and bones rather than trying to chew them into smaller portions. By contrast, many native mammal species, particularly dasyurid marsupials, comminute or masticate their food prior to swallowing (Hume 2003). When a native animal encounters the HSDV whilst gnawing on the bait it is predominantly rejected (i.e. spat out) rather than consumed. The use of a toxic pellet within the bait does not decrease the effectiveness of the bait (Hetherington *et al.* 2007) but does significantly reduce the possibility of a non-target species accessing the toxicant (see Chapter 7 of this thesis). When the HSDV is included within the Eradicat® bait media it is known as Curiosity® cat bait (Johnston *et al.* 2011). The Curiosity® cat bait is buffered to alkaline, using sodium bicarbonate, to allow stability of the acid soluable HSDV.

The Curiosity® cat bait in conjunction with PAPP encapsulated in a HSDV has great potential for delivering a toxicant to feral cats while minimising the possibility of non-target species coming in contact with the toxicant. It does not overcome the difficulties associated with cats preferring to hunt live prey over scavenging. This can be mitigated to some extent by targeting bait delivery at times when cats are food stressed. Additional research is required to assess the acceptability of the bait by the non-target species that may be able to consume the HSDV (See chapter 7). Further research is also required to determine if the use of HSDVs will increase target specificity for other pest species for which poison is the currently preferred method of population management.

Exclusion fencing: Predator proof exclusion fencing can be used to provide secure areas for native species and to prevent reinvasion of areas that have been cleared of introduced predators. Exclusion fencing is expensive to erect and must be regularly maintained to maximise its effectiveness (Moseby and Read 2006). The 11 kilometre exclusion fence surrounding Mulligans Flat Nature Reserve in the ACT cost just over $AUD 1.3 million to erect (ABC 2009). Sections of the fence were knocked over by flood water in 2010, potentially allowing reinvasion by predators while repairs were being undertaken. It is now being modified to allow flood water to pass through (ABC 2010). Exclusion fencing has been used successfully to assist reintroduction, or recovery, of endangered species in areas that they once occupied. The Arid Recovery project in South Australia (Moseby and O'Donnell 2003) and the Heirisson Prong project in Western Australia (Morris *et al.* 2004) are examples of exclusion fences that have proved successful for the reintroduction of native species. However, even with continual maintenance and monitoring of exclusion fencing, breaches are still likely to occur (Saunders *et al.* 1995).

Trap-neuter-release (TNR): There are increasing calls from within the community advocating the trapping, sterilisation and re-releasing of feral cats into the environment as an appropriate management strategy (Andersen *et al.* 2004; Levy and Crawford 2004; Schmidt *et al.* 2009). TNR programs are used extensively in North America and are advocated by cat welfare groups on the pretext of managing feral cat populations based on the assumption that TNR is more humane than euthanasia (Foley *et al.* 2005; Winter 2004). While not widely promoted in Australia, literature is beginning to appear in peer-reviewed Australian journals advocating TNR (e.g. Schmidt *et al.* 2009). Some advocacy groups argue that feral cats fill a necessary niche in the environment or do no harm to native species (for examples of these claims see Longcore *et al.* 2009). However, there is little or no evidence that TNR programs effectively manage feral cat populations, particularly while there is continued recruitment to the population through deliberate dumping of unwanted pet cats (Jessup 2004; Winter 2004). There is a large volume of scientific evidence to refute the claims put forward in support of TNR as an effective management strategy (see reviews by Lepczyk *et al.* 2010; Longcore *et al.* 2009).

Public perceptions towards feral cats

Felis catus occupies a unique position in society in being both a loved family pet and a loathed invasive predator. Feral cats are often perceived within the community as a significant threat to native wildlife and most community surveys show a high level of support for managing feral and urban cat populations (Johnston and Marks 1997; Lilith *et al.* 2006). Unfortunately, there is a perception within some sections of the community that feral and domestic cats are separate species (Dickman 1996). This view may well have been encouraged by the similarly erroneous view that feral cats are larger than their domestic relatives (Dickman 2009). These views persist despite the glut of published material, both peer-reviewed and otherwise, that dispels these myths (see Denny and Dickman 2010 for a review).

Despite the often high level of support for managing feral cats, there is also strong public affection for them (Grayson *et al.* 2002). A simplistic search of the term "feral cat" in Yahoo!® Groups (undertaken 29th December 2010) revealed 309 separate groups worldwide having an interest in feral cats (Yahoo! inc 2010). Most promote the saving, care and protection of feral cats (see TNR section above). Only four of the 309 groups advocated the management or removal of feral cats from public lands. Additionally, cats are viewed by many rural land holders as being beneficial to their farms. Stray cats are tolerated around farm sheds, hay and grain storage areas, and are seen as a cheap method of rodent control (Hamilton *et al.* 2006).

Introduced animals have been present in Australia since European settlement (Olsen 1998) and many indigenous communities now view these animals as belonging to the land rather than as invaders. The feral cat is no exception to this. In some communities the feral cat has a "Dreaming" and has been incorporated into Aboriginal law. In these communities the feral cat is now seen as part of the natural environment (Rose 1995). In other communities, the feral cat is an important food resource and is hunted in the absence of traditional native species (Olsen 1998). However, not all indigenous communities view the feral cat in such a favourable manner, especially when cats impact negatively on reintroductions of native fauna with which communities are involved (Gibson *et al.* 1995; Rose 1995).

These highly variable public perceptions and views towards feral cats have restricted the ability of various statutory authorities to implement effective programs for the management of feral cats, particularly in urban areas (Dickman 2009).

Conclusion

Feral cats have been present in Australia since European settlement. They have colonised all areas on the mainland and Tasmania and many off-shore islands. While there is little experimentally-based evidence of the level of damage that they cause to native wildlife populations, there is a large body of historical, correlative and empirical evidence that implicates them in the decline of many native species. While predation by feral cats undoubtedly occurs on native species and has the potential to exert downward pressure at a population level, many other factors can also influence population declines such as land clearing, overgrazing by domestic stock, epizootic disease, urbanisation and competition with other introduced species such as rabbits. Feral cats can severely compromise attempts to reintroduce small and mid-sized native animals, particularly if the reintroduced animal is from a captive-reared population with no previous exposure to predators. Domestic cats potentially can have greater detrimental impacts than feral cats on species that have been restricted to small remnant patches of suitable habitat through urbanisation. Domestic cats obtain all their food resources from their owners and hunt for play rather than through necessity. This allows them to put greater downward pressure on populations of small prey species as they can continue to hunt long after it would have become energetically and nutritionally unprofitable for feral cats that rely on predation for their food. There are few techniques available for the effective management of feral cat populations, particularly in Victoria where leg-hold traps now cannot be used to capture cats on public land. However, on-going research on cat ecology and behaviour in different environments and on methods of improving the target specificity of new baits and toxicant delivery techniques hold promise for the future. Aspects of feral cat ecology and attempts to improve baiting protocols are described in subsequent chapters.

Chapter 3

Home range of feral cats (*Felis catus*) in tall forests of Far East Gippsland, Victoria

Chapter overview

In this chapter, I describe the home ranges of feral cats in the tall closed forests of Far East Gippsland, Victoria and compare these with home ranges of cats in other habitat types. Most research into feral cats in Australia has focused on the arid and semi-arid regions and consequently little is known about the ecology and home ranges of feral cats in forests. Here I redress that knowledge gap.

Twenty-two feral cats were captured and fitted with either GPS collars or conventional VHF collars to determine the extent of their home ranges and range overlap. Eight cats provided sufficient home range data to reach asymptote. Home range and core area sizes were calculated using both Minimum Convex Polygon (MCP) and Kernel Density Estimate (KDE) methods. There was no significant difference in the size of the home ranges calculated from GPS data and those from VHF telemetry. Male cats were significantly heavier than female cats. There was a non significant but positive correlation between cat weight and home range size. All eight cats showed site fidelity to their home range.

Male cats had larger home ranges than females. Male cat home ranges overlapped those of females. Home ranges of adjoining female cats overlapped. The core areas of two adjoining females also overlapped. The tendency for female cats to have overlapping home ranges and core areas indicates that the female cats in Far East Gippsland are tolerant of the presence of other females and do not actively exclude them. However, this overlapping is not the same as group living observed in cats in localised areas with high food resources such as rubbish tips.

Two pelage colours were dominant among the captured cats – tabby and black. More tabby cats were captured than black cats but the difference was not significant. Only a single ginger cat was caught.

Compared with other studies in Australia and New Zealand, the home ranges of feral cats in the tall closed forests of Victoria were smaller than those of cats in the arid and alpine zones due most likely to poorer food resources in those areas. Home ranges were larger than those of feral cats living on farmland or grasslands where food resources are usually greater or more accessible.

Introduction

The domestic cat (*Felis catus*) is thought to have been introduced into countries of the Pacific region with early European sea-farers (Baldwin 1980). Early European settlers introduced the cat into Australia and they spread inland from multiple points on the coastline (Abbott 2002). The spread of feral cats across Australia was aided in no small part by the European rabbit (*Oryctolagus cuniculus*) which provided a reliable and abundant food source. In order to combat the rabbit plagues in the late 18th and early 19th centuries, domestic and stray cats were released into the Australian outback in large numbers; this contributed to their spread across the country (Abbott 2002; Rolls 1969). These cats eventually formed self sustaining feral populations that now occur over the Australian mainland, Tasmania and on many offshore islands (Dickman 1996).

Feral cats prey on a wide range of prey species (Dickman 2009; Risbey *et al.* 1999; Risbey *et al.* 2000) however their impacts are not confined to direct predation and include a number of amensal effects (Dickman 1996; Glen and Dickman 2005). For example, feral cats not only depredate western quolls (*Dasyurus geoffroii*) (Glen *et al.* 2010) but potentially compete with quolls more generally for food and den resources (Glen and Dickman 2005). Feral cats are carriers of non host-specific diseases which have been found in native wildlife, including toxoplasmosis (caused by *Toxoplasma gondii*) (Dickman 1996; Moodie 1995). Additionally, the presence of predators such as cats can depress the behaviour of prey species resulting in those species, making shorter and less frequent feeding forays from the safety of their nest or place of shelter (Charnov *et al.* 1976; Stokes *et al.* 2004). This can result in the individuals not being able to successfully forage, leading to declines in individual fitness and potentially reduced population performance (Arthur *et al.* 2004).

Feral cats generally have a greater impact on island species than on mainland species. When assessing the impact of feral cats on native species, for example Dickman (1996), and later Dickman *et al.* (2010), created a separate, higher, scoring system to reflect the increased level of impact that feral cats have on island species. Meta analyses by Salo *et al.* (2010; 2007) confirmed that the impacts of introduced predators are greater in Australia than in other parts of the world, and suggested that island faunas – even continental ones – generally are more vulnerable to novel predators.

Smith and Quin (1996) found that the abundance of feral cats in areas where foxes and rabbits were present was the best predictor of the decline of small conilurine rodents (>35 g). For all conilurine rodents, feral cats were the strongest predictor of decline when foxes and rabbits were absent. Feral cat predation on threatened native species has prompted the eradication of feral cats from islands and some peninsulas in order to provide refuges for many endangered native species in Western Australia (Algar *et al.* 2002; Morris *et al.* 2004) and South Australia (Moseby *et al.* 2009a; Moseby and O'Donnell 2003). The Stephens Island Wren (*Xenicus lyalli*) was extirpated by a single cat and her kittens brought to the island by a lighthouse keeper (Galbreath and Brown 2004; Medway 2004) while the Angel de la Guarda deer mouse (*Peromyscus guardia*) is now extinct on Estanque Island, Gulf of California, as a result of predation by a single cat (Vazquez-Dominquez *et al.* 2004).

Most research on feral cats in Australia has taken place in either the arid (for example: Burrows *et al.* 2003; Moseby *et al.* 2009b) or semi-arid regions (for example Edwards *et al.* 2001; Hilmer 2010; Jones and Coman 1982). There is only one study of feral cats in the alpine zone (Watson 2006) and a single study of feral cats in tall closed forests, which examined diet only (Triggs *et al.* 1984). No research has been undertaken on home range sizes or range overlap of feral cats in tall closed forests, despite the prevalence of cats in such habitats and their probably deleterious impacts on native species (May and Norton 1996). Knowledge of home ranges and the extent of their overlap is essential for managing the impacts of feral cats on threatened native species. It gives an indication of the intensity and spatial extent of control effort that is required, and can inform managers about the utility of specific methods such as where to deploy traps, baits or other control measures.

The main aim of this project was to answer key questions about the home range sizes and the extent of range overlaps of feral cats in the tall closed forests of Far East Gippsland, and to compare them with those of feral cats from other areas of Australia. Minor aims were to explore the general relationship between body weight and range area in male and female cats, and to compare range size estimates obtained using different methods. This knowledge can be then used to design new, or adapt existing feral cat control programs to target feral cats in a more strategic manner.

Methods

Study site

This research was conducted in the Southern Ark Project area of operation in Far East Gippsland, Victoria. It was conducted in the Drummer Poisoned Monitoring Area and the West Cann Un-poisoned Monitoring Area. See Chapter 1 for a more comprehensive overview of the Southern Ark Project and the sites utilised for this research.

Feral cat capture

Feral cats were trapped using #1.5 and #3 Victor Soft Catch® (Woodstream Corporation, Lititz, Pennsylvania; current manufacturer: Oneida Victor, Inc., Euclid, Ohio) rubber-jaw traps. Trap sets comprised either a single trap, two traps or a grid of four traps closely set with sides made from surrounding vegetation to guide the cats over the trigger plate of the trap to maximise the chance of capture. Trap sets were baited with either a meat based bait (beef or chicken meat or fish based tinned cat food) or ‘Pongo’, a scent-based lure consisting of a mixture of cat faeces, cat urine and water (Algar *et al.* 2002). Meat baits were sprayed with Coopex® (Bayer Australia Ltd) mixed with water to the manufacturer’s specifications (25 g to 2.5 l water), to prevent the bait from becoming flyblown or covered in ants and therefore less attractive to the cats (Johnston *et al.* 2009; Johnston *et al.* 2008). A Feline Audio Phonic (FAPs) audio lure (Westcare Industries, Nedlands, Western Australia) was placed at about 50% of the trap sets as an additional lure. Trapping for feral cats used in this study was undertaken from January 2007 until August 2009. Trapping to recapture collared feral cats continued until August 2010.

Traps were checked from dawn each day. Depending on the level of sedation required, captured feral cats were sedated using an intramuscular injection of either Zoletil® (5 mg/kg) or Domitor® (0.15 mg/kg) or in combination (0.05 mg/kg Domitor plus 3 mg/kg Zoletil). Sedated cats were sexed, weighed and checked for trap injuries. Cats were then fitted with either a 135 g combination GPS/VHF collar with mortality sensor or a 40 g VHF-only collar with mortality sensor (SirTrack, Havelock North, New Zealand). Mortality sensors triggered after 24 h without movement increasing the VHF transmission rate from 40 ppm to 80 ppm. Cats weighing less than 2.7 kg were not fitted with a combination collar in order to keep the weight of the collar under 5% of the body weight. GPS collars were tasked to take fixes at either a combination of: a) 15 minute intervals for 4 h then hourly intervals for 23 h; or b) 6 h intervals. Fix data were stored onboard the unit. Battery life at these tasking rates was approximately 2.5 and 5.5 months, respectively. Prior to fitting the collars, the almanac on the GPS collars was updated in accordance with the manufacturer's instructions to ensure locational accuracy. The actual VHF transmission frequency for both style of collars was established and entered into the R 1000 radio receiver (Communications Specialists, Orange, California). Collars were not fitted with a timed drop off device, consequently cats had to be recaptured to recover GPS fix data.

GPS data

GPS data were downloaded from the collars using the Sirtrack download interface in comma-separated values (CSV) format. The GPS collars store position data using the Geographic Coordinate System (Lat/Long) (GCS). Data stored this way are not compatible with many of the analysis plug-ins and extensions that are used in Geographic Information System (GIS) programs. The downloaded data were transformed to remove incompatible formatting by loading into the program DNR Garmin (Department of Natural Resources, Minnesota); this converted the projection from GCS to the Universal Transverse Mercator (UTM) system in the WGS 84 projection. These transformed data were then loaded into the ArcMap 9.2 GIS program (ESRI Inc, Redlands, California). Hawth's Analysis Tools for ArcGIS (Beyer 2004) and Home Range Tools (Rodgers and Kie 2007) plug-ins were used to determine the home range and movement patterns for each cat. ArcView 3.2 (ESRI Inc, Redlands, California) with the Home Range extension (Rodgers and Carr 1998) and the Animal Movement extension (Hooge and Eichenlaub 2000) were used to determine site fidelity, turn angle and step length and to undertake home range bootstrap analyses.

Horizontal dilution of precision

A Horizontal Dilution of Precision (HDOP) value is calculated by the GPS for each fix based on the position of the satellites in the sky that it has been able to detect and obtain data from. A small HDOP value indicates that the available satellites were spread widely across the sky and indicates a high level of precision. A large HDOP value indicates that the satellites were closely grouped in the sky, thus increasing the probability of locational error. All fixes with a HDOP value of 4 or greater were discarded when calculating home range size because of the lower level of precision achieved for these fixes (Moseby *et al.* 2009b).

VHF data

Radio telemetry triangulation of the signal from the VHF transponders was used to determine locational fixes for the cats. VHF tracking was undertaken during both the day and the night. To minimise the potential for autocorrelation in the data (Swihart and Slade 1985), a single fix was taken each day when tracking during daylight hours. Timing of the fix varied on each day to avoid potential bias arising from cats returning to daytime dens. As cats were more active at night, two fixes were taken per night when night time radio tracking. Night time fixes were taken at least 6 hours apart. No daytime fixes were taken while night time tracking was being undertaken. Between 30 and 50% of fixes were taken during the night.

A single fix for a feral cat was determined from at least three individual bearings taken within a 10 minute period to minimise the errors arising from movement by the cat between successive bearings. Movement by the cat within this period can effect the accuracy of the fix (Schmutz and White 1990). It was not possible to take bearings simultaneously from three separate locations, therefore this time frame was selected as a balance between distance needed to be moved between successive bearings to minimise triangulation error and the potential for movement by the cat. The GPS coordinates where each bearing was taken were recorded together with the compass bearing of the strongest signal. The compass bearing of the strongest signal was taken three times at each location and the mean bearing was used in fix calculations. Mean compass bearings were then converted using the magnetic declination from magnetic to grid based for analysis. Magnetic declination in Far East Gippsland is +13 degrees (Geoscience Australia 2007). The strength of the VHF signal as shown on the receiver (0 = very low to 5 = very strong) was noted. Where possible, the angle between successive bearings was between 30 and 60 degrees to minimise the potential error in location that can occur when intersection angles are too great or too small. When the angle between successive bearings was greater or less than this, the error polygon created in Locate III Radio Telemetry program (Pacer Computing, Canada) was examined. Decision to retain or exclude the fix were made on a case by case basis taking into account the distance between the point at which bearing was taken and the probable location of the cat, the potential for signal bounce from the surrounding topography and the strength of the signal for each bearing. Radio tracking data were entered into the Locate III program to determine the location of the cat. These data were then exported in ArcView format and imported into the GIS for further analysis.

Home range calculation

The home range of each cat was calculated using the 100% Minimum Convex Polygon (MCP 100) estimator (Mohr 1947) as home ranges calculated using MCP are comparable between studies whereas many of the alternative techniques are not (Harris *et al.* 1990; Kernohan *et al.* 1998). Forays outside the home range were removed prior to estimation of home range size. Forays were assessed using the continuous asymptote function of the Abode plug-in (Laver 2005) in ArcGis 9.2. Spikes (large increases) in the home range size were examined as possible roaming events. Date points were excluded if that point had no others nearby to indicate common usage of the area. By convention, roaming outside of the home range area on occasional forays should not be included as part of home range estimates (Burt 1943).

Home range estimation using Kernel-density estimation (KDE) (Kernohan *et al.* 1998; Worton 1989) was also undertaken. This method has the advantage over simple area-based estimates such as MCP in that it is probabilistic and not unduly biased by the inclusion of occasional forays or roaming events. The fixed kernel estimator was used with a smoothing factor calculated using the least square cross validation (LSCV) method (Worton 1989). Output cell size was set at 10 m^2 in order to avoid potential bias through using larger cell sizes (Laver 2005). One cat, Hans, had two fix rates tasked for the GPS collar. To avoid potential bias when determining the density of usage within the home range (i.e. more fixes in certain areas due to a more rapid fix rate), only those fixes taken at least an hour apart were used for KDE analysis. Home range size was calculated using the 95% density isopleth (KDE 95), thus removing outliers and further down-weighting the potential effects of forays and conforming with usage in previous works (e.g. Molsher *et al.* 2005)

Home range overlap

The intersect tool in ArcGis 9.3 was used to determine the extent of any overlap of the MCP 100 home ranges of adjacent cats. The size of overlap was calculated using the Table Tools function in Hawth's Analysis Tools for GIS. The degree of overlap is expressed as a percentage of the MCP 100 home range of each overlapping animal (Genovesi and Boitani 1997).

Core area

The core area of the home ranges was calculated based on the 50% of points closest to the harmonic mean centre of the home range (MCP 50) (Molsher *et al.* 2005) and the 50% density isopleth using KDE (KDE 50). The harmonic mean centre in MCP analyses allows the centre point of the home range to move depending on which points are removed during the analysis. Core area calculations for the cat Hans used only fixes taken at hourly intervals to avoid bias that may be introduced by including the higher fix rate data.

Autocorrelated data

All tracking data are autocorrelated, but this is especially so for GPS data due to the regularity of fixes when using the GPS collars. However, this was not considered to pose a large problem for analysis or interpretation as both the MCP and KDE techniques of calculating home range are robust with respect to autocorrelated data (Kernohan *et al.* 2001). The negative bias introduced into home range estimates is minimal when using MCP and KDE analyses. Swihart and Slade (1997) also found that the use of non-autocorrelated data is not necessary when estimating home ranges using those methods.

Asymptote

The MCP estimator is sensitive to sample size, with home range size increasing proportionally to the sample size until an asymptote is reached. No further increase in home range occurs once asymptote is reached regardless of further increases in sample size (Laver 2005). It is not possible to accurately estimate home range sizes unless asymptote has been approached. Spikes in home range size observed during asymptote analysis generally indicate either a range shift or foray outside the usual home range (Laver 2005). These data points were identified and removed from the analysis if determined to be a foray, rather than a range shift outside the home range. Asymptote was calculated using two separate techniques:-

a) the asymptote analysis function in Abode for ArcGis 9.2 employing the random selection of points technique. Random selection was used as the data were discontinuous following the removal of foray data points and those fixes with a HDOP >4 (Harris *et al.* 1990; Laver 2005). An asymptote was considered to have been reached when there was less than 5% variation in home range size for at least 5 consecutive fixes for VHF telemetry or for at least 10% of all fixes for data gathered using a GPS; and

b) the MCP Bootstrap function of the Animal Movements extension in ArcView 3.2 (Hooge and Eichenlaub 2000; Laver and Kelly 2008), using 100 replicates for each analysis. Asymptote was deemed to have been reached when the 95%

confidence interval of the home range estimate was within 5% of the total home range estimate for at least 5 consecutive fixes for VHF telemetry and for at least 10% of all fixes for the GPS data (Harris *et al.* 1990; Laver and Kelly 2008).

Home ranges were considered to have attained asymptote only if both techniques indicated asymptote had been reached.

Site fidelity

Site fidelity can be used to quantitatively determine the existence of a home range. An animal that exhibits site fidelity is deemed to possess a home range (Spencer *et al.* 1990). It exists if the area used by an animal is significantly smaller than the area used if an animal moved at random (Munger 1984). Site fidelity of feral cats was tested using Home Range Extension in ArcView 3.2. This analysis compared the home range of the feral cat against that created by 100 random walks based on the turn angles and step lengths of each individual cat. The random walks all commenced at the first data point for each cat. The mean square displacement (MSD) of each walk was calculated from that point and compared with the MSD of the actual movements of the cat.

GPS collar fix rate

GPS collar fix rates were calculated using the expected number of fixes that would be obtained from the tasking rate compared with successful fixes by the GPS collars. The time of last fix was calculated for cats that died during the study as the time of the last fix prior to any fixes taken of the point where the collar was located. Fixes taken at that point were deemed to have occurred after the cat died and were discarded. Where the GPS battery on the collar had become flat, the last fix with a HDOP <4 was used at the end point.

General analyses techniques

Comparisons between male and female home range sizes and between home ranges determined through GPS or VHF telemetry were made using the Mann-Whitney U test. This test was used due to the small sample size obtained and the non parametric nature of the data. Comparisons between male and female cat weights were made using a single factor ANOVA. Spearman correlation analysis was used to determine the relationship between cat weight and home range size. All analyses were undertaken using the Statistica 7 (StatSoft Inc 2004) statistical analysis package.

Results

Twenty-two feral cats were captured during this study. Eleven were fitted with combination GPS/VHF collars and a further six were fitted with VHF-only collars. Two cats captured outside the study area were fitted with reflective collars for remote camera detection, one escaped prior to being fitted with a collar and two others were humanely euthanised for dietary analysis (Table 4). Three of the collared cats died and the collars were located with the remains of the cat. An additional collar was located after the mortality sensor activated, but the remains of that cat were not found. This collar bore marks consistent with being chewed by a fox or dog. It is unknown if this cat died as a result of being depredated or if it died and was subsequently eaten as carrion. Five cats moved into inaccessible areas not long after being collared and were unable to be located regularly using the VHF receiver. Only three cats were recaptured despite an extensive trapping effort. The remaining cats were monitored using VHF telemetry until the battery in the transmitter went flat and they were no longer able to be detected. Data sufficient to provide meaningful results were obtained from eight cats.

Sexual dimorphism and pelage

Male cats were heavier than female cats ($F_{1,19} = 17.9$, $P < 0.001$). A greater number of cats with tabby pelage were captured (n = 13) compared to black (n = 8) or ginger (n = 1) but the frequency difference between tabby and black pelage was not significant (= 0.76, P ns). Overall, there was no difference between the weights of cats with tabby or black pelages ($F_{1,18} = 0.3$, $P = 0.56$) or between sexes with different pelage colours ($F_{1,16} = 0.01$, $P = 0.90$; note that the cat with the ginger pelage was excluded from that analysis).

Table 4. Cats captured during research. * = cat died, ** = cat recaptured, # = collar recovered ## = cat disappeared with limited data gathered, ♣ = that cat was not recaptured and no GPS data were gathered; some VHF data gathered

Cat	Weight (kg)	Sex	Pelage	Collar fitted
Liz ♣	3.4	F	Tabby	GPS/VHF
Chris ♣	5.2	M	Tabby	GPS/VHF
Karen # *	2.9	F	Tabby	GPS/VHF
Robyn ## ♣	3.3	F	Black	GPS/VHF
Dave ## ♣	3.9	M	Black	GPS/VHF
Olof # **	4.6	M	Tabby	GPS/VHF
Mike # *	5.6	M	Tabby	GPS/VHF
Neil # **	4.2	M	Tabby	GPS/VHF
Brian ## ♣	6.2	M	Black	GPS/VHF
Hans # **	4.3	M	Tabby	GPS/VHF
Grant ♣	6.0	M	Tabby	GPS/VHF
Danielle	3.9	F	Tabby	VHF
Hayley	3.4	F	Black	VHF
Sue	3.5	F	Black	VHF
Steffi` ## *	3.5	F	Tabby	VHF
vhf 34 ##	5.1	M	Tabby	VHF
James	4.8	M	Ginger	VHF
Euthanised	2.7	M	Tabby	Nil
Euthanised	3.0	F	Black	Nil
Reflective collar	3.6	F	Black	Nil
Reflective collar	5.1	M	Black	Nil
Escaped	n/k	M	Tabby	Nil
Mean male (± se)	4.8 ± 0.2			
Mean female (± se)	3.3 ± 0.1			

GPS collar fix rate

The duty cycle of the collars was set at the time of manufacture and was not able to be changed in the field. The mean time between fixes was less than the fixed amount requested (Table 5). This resulted in a slight backward shift in time for each subsequent fix. Fix success rate varied between collars (23.5% to 71.3%) with a mean success rate of 49.2%. The area the collar was operating in had no appreciable effect on fix success as separate collars on the same cat had both the highest (71.3%) and the lowest (23.5%) success rates (Table 5). Each cat also showed extended periods when no fixes were obtained.

Table 5. Fix rate and fix success rates for the GPS collars on feral cats

Cat	Duty cycle time between fixes	Mean time between fix attempts ± se	Fix rate
Olof (1st collar)	6 h	5.93 ± 0.02 h	71.3%
Olof (2nd collar)	6 h	5.96 ± 0.004 h	23.5%
Karen	6 h	5.97 ± 0.001 h	51.3%
Neil	6 h	5.97 ± 0.001 h	38.0%
Hans	1 h	59.44 ± 0.00002 mins	61.9% total
Hans	15 mins	14.54 ± 0.00003 mins	
Mean fix rate ± se			49.2 ± 8.5%

Site fidelity

All cats showed site fidelity, with each exhibiting more constrained movement than each of the 100 random walks ($P<0.01$) generated using the Animal Movements extension for ArcView 3.2 (Figure 7). This indicates that the cats were constrained within the boundaries of a home range, with the exception of occasional forays outside that area.

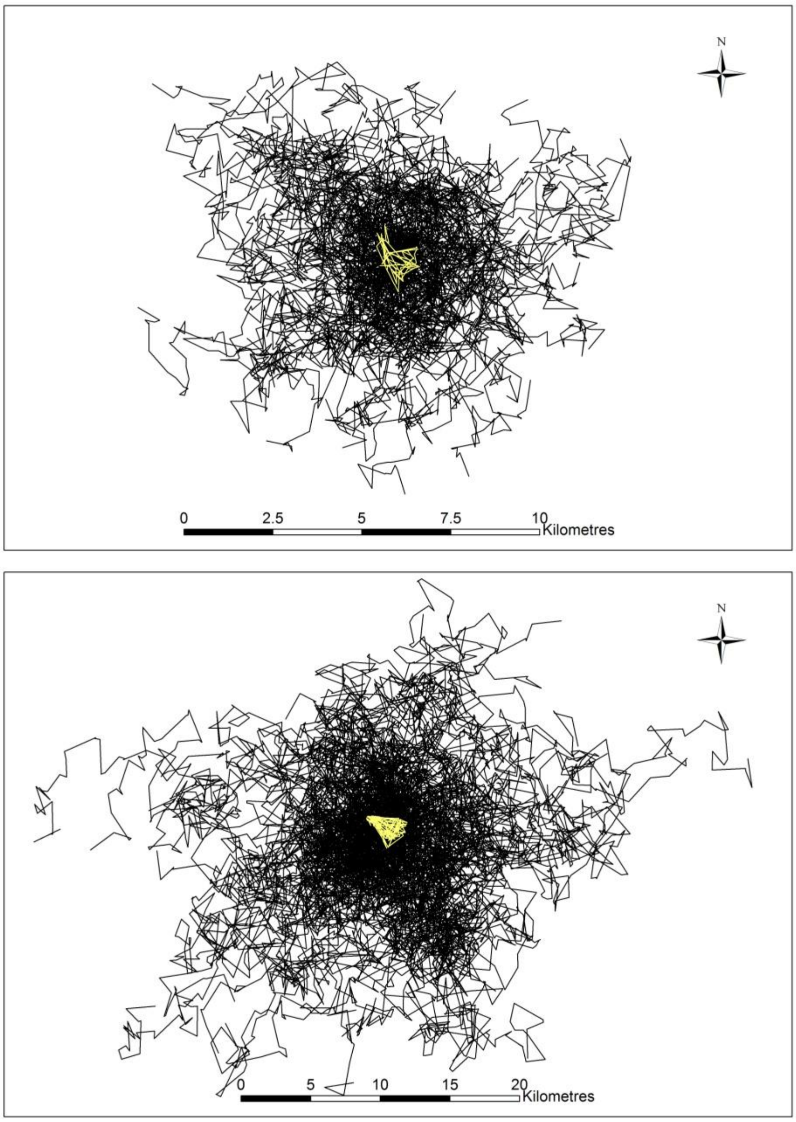


Figure 7. Site fidelity diagrams for two feral cats. The dark lines are randomly calculated walks taken from the distribution of step length and turn angle for each cat. The light line at the centre is the actual movement patterns of the cat. All cats exhibited site fidelity, however only two are shown as examples.

Asymptote

Each of the GPS collars that were recovered reached home range asymptote and two of the cats fitted with GPS collars that were not recovered reached asymptote using the fixes obtained from VHF telemetry. A further two cats fitted with VHF only collars also reached asymptote (Figure 8; Table 6). Asymptote was not reached using either method for the remaining cats, and these cats have been excluded from further analysis.

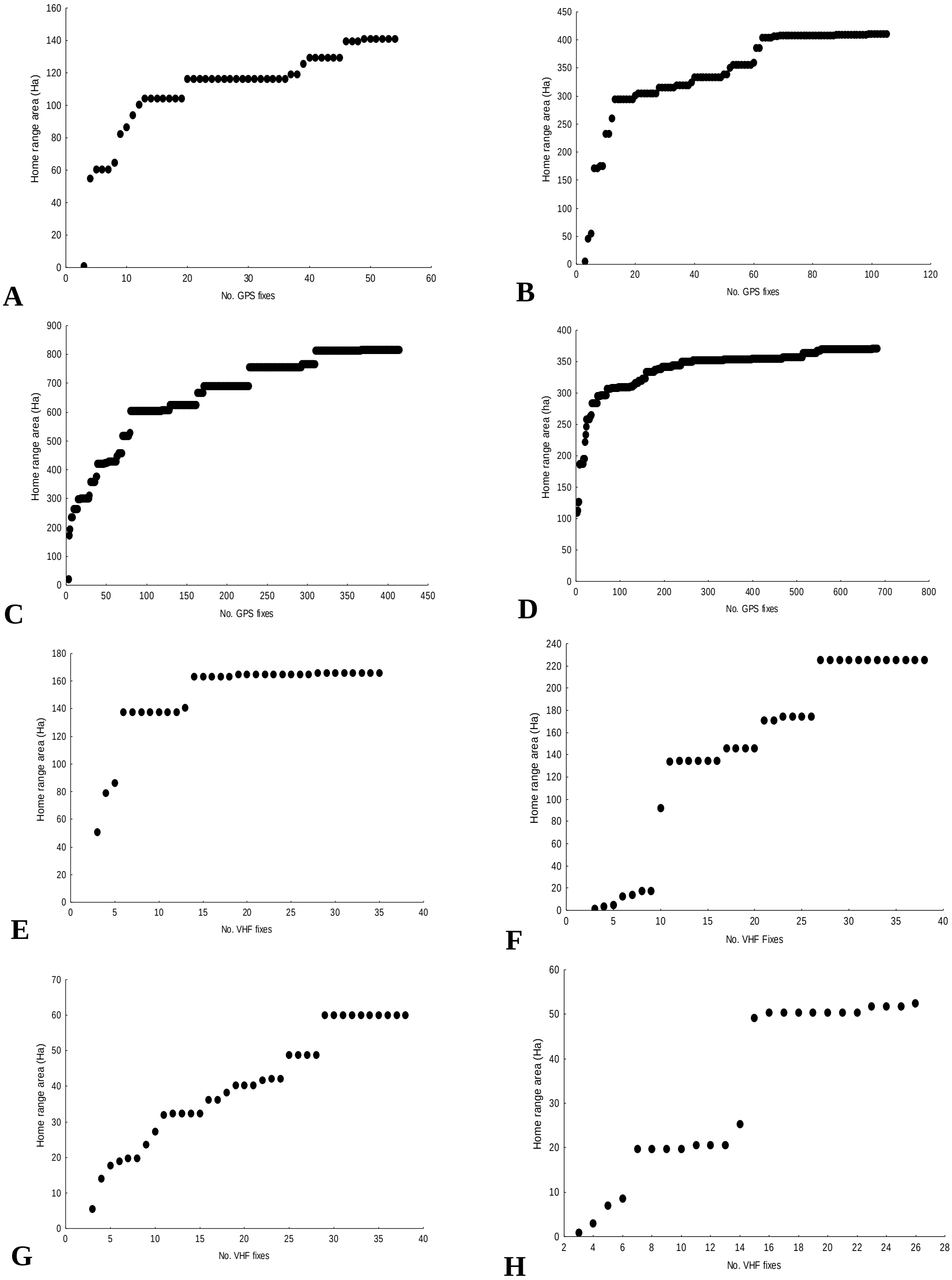


Figure 8. Asymptote graphs for MCP home range for cats. Graphs A-D are for data collected via GPS collars and graphs E-H are for those collected using VHF telemetry. A-Karen, B-Neil, C-Olof, D- Hans, E-Liz, F-Chris, G-Danielle, H- Hayley.

Table 6 Number of fixes required to reach asymptote for each feral cat using 2 separate methods

Cat	Fixes to reach asymptote	
	Method 1 [a]	Method 2 [b]
Olof	307	305
Neil	61	61
Karen	44	45
Hans	163	115
Liz	12	30
Hayley	14	20
Danielle	27	29
Chris	25	32

[a] Analysis using ABODE using random selection of points
[b] Analysis using MCP bootstrap method in ArcView 3.2 using 100 iterations

Home range

MCP 100

The MCP 100 home range varied from 53 to 816 ha with a mean male home range (± se) of 455 ± 126 ha and mean female home range (± se) of 105 ± 28 ha (Table 7). These ranges fall within the median home range size for other studies undertaken in Australia and New Zealand using non-urban feral cats (Table 8).

Male home ranges were larger than female home ranges (Mann-Whitney *U* test Z = -2.3, *U* = 0, *P* = 0.02). Home ranges determined by GPS were not significantly different from those determined by VHF telemetry (Mann-Whitney *U* test Z = 1.7 *U* = 2, *P* = 0.08). Core areas, as defined by MCP 50, were larger for males than for females (Mann-Whitney *U* test Z = -2.0, *U* = 1, *P* = 0.04). A positive correlation existed between cat weight and home range size but this was not significant (Spearman’s R = 0.68, *P* = 0.06).

Table 7 Home range of cats collared in this study. * indicates the number of points used in the KDE analysis. See text for explanation.

Cat (sex)	Period tracked	Collar type	MPC 100 (ha)	MCP 50 (ha)	KDE 95 (ha)	KDE 50 (ha)	Points used
Karen (F)	6 wks	GPS	141	42	179	43	54
Neil (M)	5 mths	GPS	410	162	546	52	105
Hans (M)	3 mths	GPS	370	131	446	41	683/203*
Olof (M)	9 mths	GPS	816	184	595	85	414
Liz (F)	19 mths	VHF	166	12	137	16	34
Chris (M)	18 mths	VHF	226	33	246	21	37
Hayley (F)	11 mths	VHF	53	5	58	11	25
Danielle (F)	14 mths	VHF	60	21	90	17	37
Mean ± se			280 ± 89	73 ± 25	287 ± 74	37 ± 9	
Mean Female ± se			105 ± 28	20 ± 8	116 ± 26	22 ± 7	
Mean Male ± se			455 ±126	127 ± 33	458 ± 77	49 ± 13	

Kernel analysis

KDE 95 home ranges for males were larger than those for females (Table 7) (Mann-Whitney *U* test Z = -2.3, *U* = 0, *P* = 0.02). Unlike with MCP analysis, home ranges determined by VHF telemetry were significantly smaller than those described by GPS (Mann-Whitney *U* test Z = -2.0, *U* = 1, *P* = 0.04). As with MCP home ranges, a positive trend existed between cat weight and home range size, but this was not significant (Spearman's R = 0.65, *P* = 0.08)

Core area

Using the MCP 50 method for core area calculations for cats with GPS collars resulted in the inclusion of areas that cats apparently did not use as part of the core areas. Furthermore, this method also resulted in some areas of high use being excluded from the calculated core area (Figure 9). These problems did not occur to the same extent when using KDE 50 analysis. Additionally the KDE analyses indicated multiple core areas for the cats rather just a single area, as did the MCP 50 method. Core areas calculated using MCP 50 were significantly larger for males than females

(Mann Whitney *U* test Z = -2.0, U = 1, *P* = 0.04), however using KDE 50 analysis, core areas for males were not significantly larger than those for females (Mann-Whitney *U* test Z = -1.7, U = 2, *P* = 0.11).

Table 8 Comparison of home ranges of VHF and GPS tracked non urban feral cats in Australia and New Zealand in order of increasing male home range size. * indicates home range is assessed from observation rather than telemetry. ** indicates that 95 MCP was used so is smaller than would be found using MCP 100 analysis. ++ indicated GPS rather than VHF telemetry used.

Study	Habitat / Location	Female (ha)	Male (ha)
Brothers *et al.* (1985)	Grassland / Macquarie Island		41*
Fitzgerald and Karl (1986)	River Valley / New Zealand	80	140
Schwarz (1995)	Open forest / Australia	29	154
Norbury *et al.* (1998)	Farm land / New Zealand	249	189
Moller and Alterio (1999)	Coastal grassland / New Zealand	167	207
Langham and Porter (1991)	Farm land / New Zealand	124	240
Molsher *et al.* (2005)	Open woodland / Australia	238	426
Gillies *et al.* (2007)	Forest / New Zealand	117	446
This study ++	**Tall closed forest / Australia**	**105**	**455**
Jones and Coman (1982)	Semi-arid / Australia	170	620
Watson (2006)	Alpine / Australia	409	628
Recio *et al.* (2010) ++	River valley / New Zealand	1607	876
Hilmer (2010) ++	Open bushland / Australia	720	1120
Burrows *et al.* (2003)	Arid desert / Australia	700**	1200 - 1500**
Harper (2004)	Wetland valley, New Zealand	1109	2083
Edwards *et al.* (2001)	Semi-arid woodland / Australia		2210
Moseby *et al.* (2009b)++	Arid rangelands /Australia	2078**	3232**

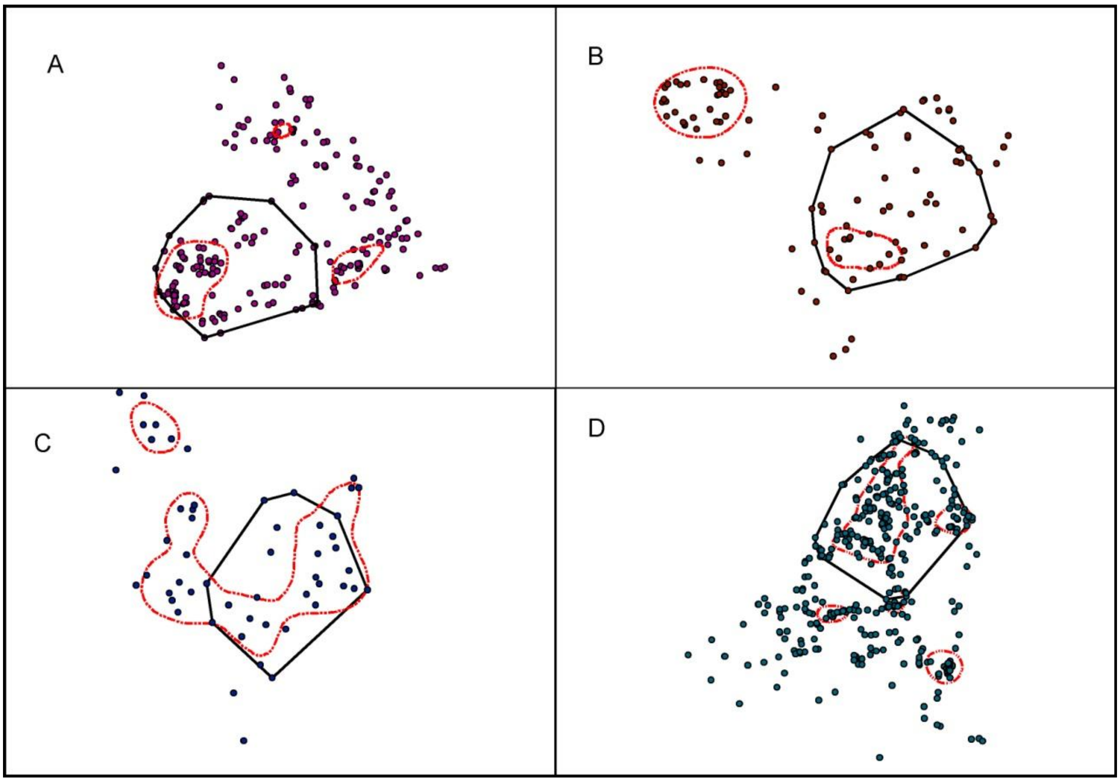


Figure 9. Core area for each cat fitted with a GPS collar. Solid line is the core area described using MCP 50. Red dotted line is that described using KDE 50. A- Hans B – Neil, C – Karen, D – Olof .

Home range overlap

Four of the cats that were tracked using VHF telemetry had home ranges that overlapped with a neighbouring feral cat. The degree of overlap varied between cats with one female (Hayley) having her home range incorporated completely within that of a male (Chris). The home range of one female (Danielle) overlapped with that of the female Liz and with that of the male Chris. The home range of Liz also overlapped the home range of another adjoining female (Sue) (Figure 10,Table 9). The core area of the male cat Chris overlapped with that of the female Hayley by 4.6 ha or 21.9 % and 41.8% respectively of their core ranges. The MPC 100 home range for Chris overlapped the core area of the female Danielle. The core area for the two females, Liz and Sue overlapped by 3.1 ha or 18.0 % and 44.9% respectively of their KDE 50 core areas. No cat for which a GPS collar was recovered had a home range overlapping with another cat.

Table 9. MCP 100 home range overlap between feral cats.
*** The home range area of the cat Sue did not reach asymptote, as a result the volume and percentage of home range overlap is calculated based on the known home range area rather than the full area.**

Cats	Area of overlap (ha)	Percentage of home range
Chris / Hayley	53	23.5 / 100
Chris / Danielle	2.2	1.0 / 3.6
Liz / Danielle	2.6	1.6 / 4.3
Liz / Sue*	37.7	22.7 / 60.9*
Mean ± se	20.3 ± 10.4	27.1 ± 12.6
Mean F / F overlap ± se		32.1 ± 16.4

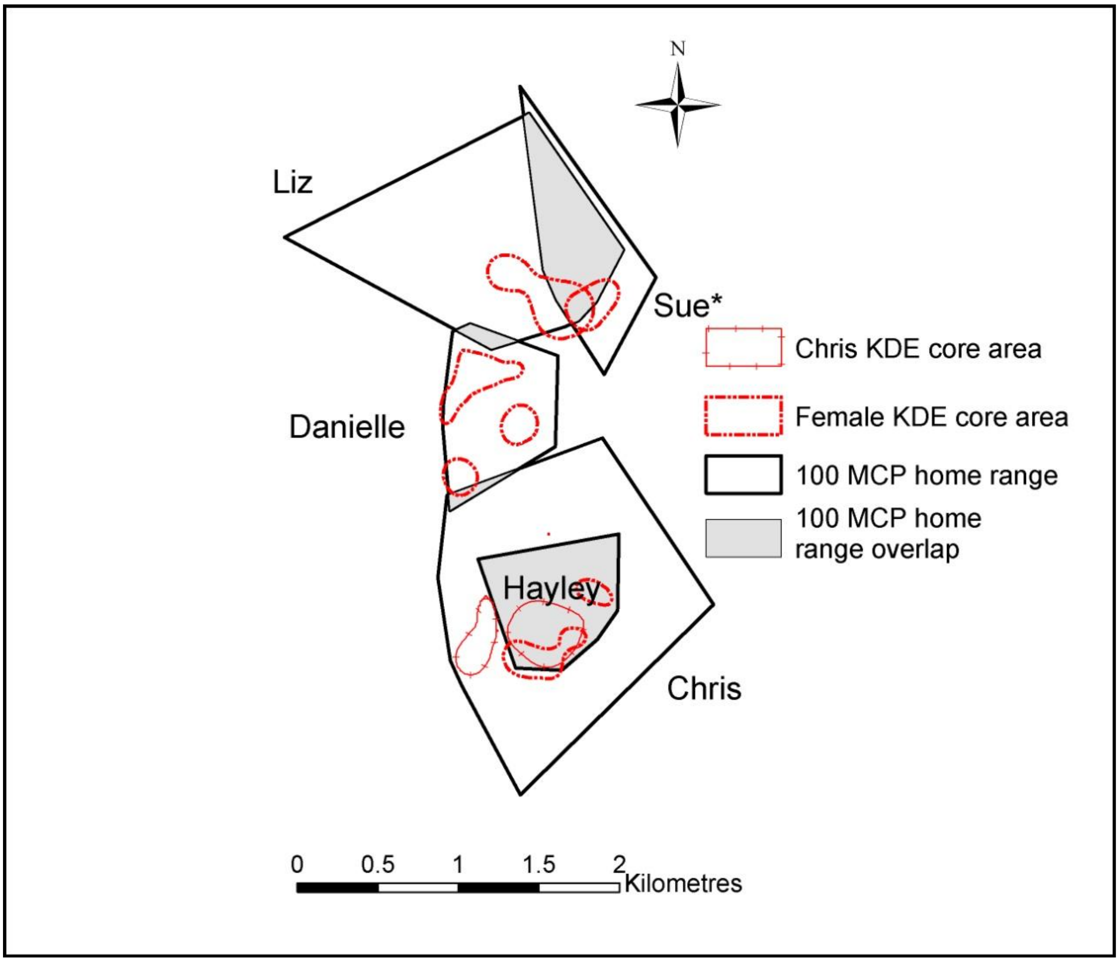


Figure 10. Home range overlap of radio tracked feral cats. * The home range for the cat Sue did not reach asymptote so was not fully described and core area was not determined using KDE. The extent of the overlap between cats Liz and Sue is based on the known volume of Sue's home range rather than the potential full volume.

Unused areas in the home range

The volume of fixes and regular fix rate of the GPS collars allowed a clearer assessment of intra home range use by the feral cats than is possible with conventional VHF telemetry. Each of the home ranges described for feral cats had areas (or holes) that were not used by the feral cats during the time the GPS collars were active (Figure 9).

Discussion

This is the first study of the home ranges of feral cats in tall closed forests in Australia using either VHF telemetry or GPS receivers. VHF telemetry has been used in other habitat types for feral cats (Table 8) while GPS collars have been utilised on other feral cat populations on islands (Dirk Hartog Island - Hilmer (2010), Johnston *et al.* (2010b) and French Island – Johnston *et al.* (2008)) and mainland sites (Alcoa Lease site, Anglesea, Victoria - Robley *et al.* (2008) and Arid Recovery, South Australia - Moseby *et al.* (2009b)). Compared with other studies in Australia and New Zealand the home ranges of feral cats in the tall closed forests were smaller than those of cats in the arid and alpine regions. This is due most likely to fewer or poorer quality food resources in those latter areas. Conversely, home ranges were larger than those of feral cats living on farmland or in grassland habitats where food resources are usually greater or accessibility to food is easier.

There is a general correlation between carnivore home range size and food resources in which low food resources result in large home ranges and abundant food resources tend to result in smaller home ranges (Sandell 1989). The availability of food resources was found by Liberg *et al.* (2000) to be one of the main factors determining the home ranges of feral cats. Those living on rich clumped food resources, such as rubbish tips and feeding stations, had very small home ranges while the largest ranges were found in feral cats living on highly dispersed natural prey items. The large home ranges found for feral cats in arid and semi-arid areas are thought to be the result of scarce and dispersed prey (Burrows *et al.* 2003; Edwards *et al.* 2001). While no sampling was conducted of prey availability in the home ranges of cats in the present work, all captured cats were in good condition with obvious deposits of body fat, indicating that the animals presumably had access to reliable food resources.

Site fidelity was clearly demonstrated by the feral cats in this study with each cat having a defined area where it remained unless undertaking short-term forays. The area used by each cat was more constrained that would be expected if the cats were moving randomly through the forest (Hooge and Eichenlaub 2000; Munger 1984). Site fidelity exists when the observed area used by an animal is smaller than the area that would be used if the movements are random. An animal is deemed to possess a home range if it exhibits site fidelity (Spencer *et al.* 1990).

Sexual dimorphism was apparent in home range sizes with males having larger home ranges than females. This disparity is apparent in all studies of non-urban feral cats with the exception of the work of Recio *et al.* (2010) (see Table 8). However, only a single female cat was used in that study so it may not be representative of the true population mean. Liberg *et al.* (2000) reviewed 28 studies of feral cat home ranges and found that male cat home ranges are generally 3 times as large as those of females. In this study male home ranges were over 4 times larger than those of females. This could reflect the considerably larger body sizes of the males compared to the females that were tracked (Table 4), but could equally be the result of low sample sizes.

The trend towards larger home ranges for heavier cats has been found in a variety of habitat types. Molsher *et al.* (2005) found that heavier cats had significantly larger home ranges in open woodland and Page *et al*. (1992) found similar increases in home range size with weight for cats at the Avonmouth docks. This increase in home range is presumably because of the need for higher energetic and nutritional intake (Nagy 2005; Nagy *et al.* 1999) and therefore a correspondingly greater prey volume intake. In addition, larger ranges would be advantageous to males in allowing them to overlap the ranges of multiple females and hence gaining priority access to females during the breeding season (Kirby and Macdonald 1988).

Generally felids are solitary animals and congregate in groups only when there are sufficient resources to eliminate the need to compete for food, shelter and mates

(Creel and Macdonald 1995). Only adult male lions and cheetahs predominantly live together in groups and only adult female lions and domestic cats live in groups; all other felids are asocial (Macdonald *et al.* 2000). Feral cats tend towards group living when there are high levels of rich clumped food resources such as at rubbish tips (Denny 2005; Hutchings 2003) or where humans provide food (Natoli and De Vito 1991; Say *et al.* 1999) but are solitary when food resources are scarcer. The home ranges of male cats generally overlap those of females but males generally exclude other males from their home ranges (Liberg 1980; Liberg *et al.* 2000). Conversely, Hilmer (2010) found a high degree of overlap of both home range (72%) and core area (81%) for male cats on Dirk Hartog Island and suggested that this may have resulted from a high level of prey availability in the areas inhabited by those cats. Molsher *et al.* (2005) found a lesser degree of home range overlap (21%) and core area overlap (7%) and that range overlap was not dependent on the sex of the cat.

Whilst the cats in this study appeared to be largely solitary, there was a degree of overlap of home ranges both between adjoining females and between females and males. Because it is possible also that other uncaptured and uncollared cats were present in the study area, the overlap described here should be regarded as minimal. Male home ranges regularly overlap those of females (as reviewed by Liberg *et al.* 2000; Macdonald *et al.* 2000), as was also found to be the case in this study. Adjoining females had a higher degree of home range overlap than found by Molsher *et al.* (2005).

The overlap of core areas of the two females, Liz and Sue, indicates a high level of tolerance at least between these individuals. The cat Sue was initially trapped within 10 metres of a log that the other cat, Liz, had been living in for a period of 2 months – probably raising a litter of kittens. This level of tolerance has been noted in other studies, particularly those where cats have a high level of food resources. Denny *et al.* (2002) found that this tolerance usually reflected a relatedness between the cats, but this possibility could not be tested in this study. If it is assumed that not all cats in the study area had been trapped and that food resources in the study area were insufficient to allow the kind of group living observed in cats at resource-rich sites such as rubbish tips, the overlapping home ranges of the female cats studied here suggest that female cats in the tall forests of Far East Gippsland are tolerant of each other in the same area.

If this is correct, females in the tall forest environment may be more tolerant of each other than has been reported in most other habitats. Such tolerance could be facilitated by the high structural complexity of the Gippsland forests; in the same way that high habitat complexity increases the ability of prey to escape detection by predators (Arthur *et al.* 2004; Sinclair *et al.* 1998). Complex habitat structures such as logs, litter and different densities of understorey cover could allow potentially intolerant females to avoid each other.

The variation in home range size attained through using MCP analysis compared with KDE analysis (Table 7) is likely a result of the smoothing factor that was used. Using the LSCV method for selecting the smoothing factor allows for a consistent method of analysis and comparison between studies using the same method (Laver and Kelly 2008). However, with larger data sets such as those provided by GPS telemetry compared with VHF telemetry, Hemson *et al.* (2005) found that KDE analysis became unreliable when using LSCV. The use of KDE analysis was similarly rejected by Hins *et al.* (2009) and Recio *et al.* (2010) when using larger GPS-sourced datasets. However, Laver and Kelly (2008) were satisfied with the ability of KDE analyses to handle larger datasets and recommended the use of this approach over MCP analyses. It is most likely that this variation results from the presence, or absence, of multiple data points along or near the boundary of the home range. The KDE analysis searches for other points within the radius of the smoothing factor (Laver 2005). Multiple points along the edges of the home range allows the analysis to allocate a heavier weighting to cells outside the home range than would occur at a lower density of points on the edge, thus increasing the home range size.

I found that using MCP 50 to define core area was unreliable in that it includes areas not used by cats and discards areas that cats return to frequently (see Figure 9). Harris *et al.* (1990) suggested that determining core areas may be useful for understanding intraspecific patterns of home range use. As an example, Harris *et al.* (1990) showed that the KDE 95 home ranges of adjoining male muntjac deer (*Muntiacus reevesi*) overlapped, but the core area, defined as the KDE 50 of the home range, did not. This indicated that the core areas were mutually exclusive while other deer were tolerated in the outer areas of the home range. In this study, using KDE 50 core areas did not include the large areas of the home range that were not used by cats as was suggested using the MCP 50. Additionally, the KDE analysis showed multiple core areas for most cats, indicating differing areas of high use within the home range. These areas of high use probably reflect focal points of high food and shelter resources within the home range (Harris *et al.* 1990; Lair 1987) and result in the identification of more biologically significant core areas than are uncovered using MCP analyses. As a result, the use of MCP 50 to define core area should be discouraged and the KDE 50 analysis used instead.

A number of studies have used an arbitrary cut-off point for the number of fixes needed to fully describe a home range. This cut-off point is usually taken for the minimum number needed as reported in the literature (see the review of home range studies by Laver and Kelly (2008)). The variation I found in the number of fixes required for a home range to reach asymptote indicates that such a practice is likely to result in bias of the home range size through the inclusion of home ranges that have not been fully described. This is especially so for VHF telemetry. It is recommended that asymptote analysis be conducted for all home range studies.

As GPS technology has become more readily available for use with small animals, an increasing number of studies have used this technology in preference to conventional VHF telemetry (Cain *et al.* 2005; Recio *et al.* 2010; Rodgers 2001). The GPS data-logging collars used here provided a large and unique dataset of locations. The large number of data fixes obtained using the GPS collars far exceeded that which has been possible to obtain previously using VHF telemetry alone. For example, Molsher *et al*. (2005), using VHF telemetry, obtained a maximum of 65 fixes for one cat with a mean of ~ 40 useable fixes per cat over a period of 4 months. The GPS collars used in this study based on a 6 hourly tasking rate and taking into account a mean 49% fix rate, were able to take ~ 230 fixes per cat without the researcher needing to be continually in the field. An advantage of VHF telemetry over the store on board collars is the immediate availability of the fix data whereas the cats need to be recaptured to obtain the GPS data. This problem can be overcome by the use of automated drop-off collars; unfortunately this option was not available for the collars used when my study commenced, and my intensive efforts to recapture all the cats that had been fitted with GPS collars met only with partial success.

The GPS fix rate for this study was low compared with that obtained in other studies. Recio *et al.* (2010) obtained a fix rate of 62.7% for feral cats in the Tasman Valley, New Zealand. Cain e*t al.* (2005) reviewed some 35 studies using GPS collars on larger animals and found a mean fix rate of 76% with a strong inverse correlation between the fix interval time and the fix success rate. Cain *et al.* (2005) and other researchers using GPS collars (e.g. D’Eon and Delparte (2005), DeCesare *et al.* (2005), Jiang *et al.* (2008) and Lewis *et al.* (2007)), noted that many factors can influence fix success including the type of habitat through which the animal is moving, the location and angle of the GPS antenna at the time the fix is attempted, the location of satellites in the sky and which satellites that the GPS is able to connect with. Hence, caution is warranted when comparing fix success rates between studies and between individual collared animals. While not tested in this study, it is probable that individual GPS receivers have varying ability to take a successful fix even in the same circumstances. Two separate GPS receivers used consecutively on the same cat (Olof) gave different rates of successful fixes while the home range and movement pattern of this cat did not noticeably change. The periods of missed fixes for each collar may have resulted from the cat being in a shelter at the time the fix was being taken due to the antennae not having clear access to the satellite signals (D'Eon and Delparte 2005; Moseby *et al.* 2009b).

Knowledge of home range size and overlap can assist in determining the amount of effort that is needed in a control program. Where there are large individual home ranges with little overlap, the density of animals is low. Conversely, when home ranges are smaller and there is a high degree of overlap of home ranges, density of animals is higher (Liberg *et al.* 2000). The home ranges calculated in this study were smaller than those found in the arid and semi arid regions (e.g. Burrows *et al.* 2003; Edwards *et al.* 2001; Hilmer 2010). This, together with the high degree of home range overlap, indicates that a higher bait density may be required to manage cats in tall forested habitats compared with that required in arid or semi-arid regions. Additionally, population management through trapping or shooting would require higher effort per unit area than is required for cats in arid regions.

Similarly, knowledge of home range size and, from that, cat density estimates can assist in determining if reintroduction attempts for native species are advisable. Christensen and Burrows (1995) found that cats even at low densities caused the failure of reintroduction attempts for burrowing bettongs (*Bettongia lesueur*) in the Gibson Desert. Densities of feral cats as low as 1 / km^2 have caused the failure of both bettong and bilby reintroductions outside the predator exclusion fencing at Arid Recovery in South Australia (K. Moseby, Arid Recovery, pers. comm.).

GPS collars were found to allow more location fixes to be taken per animal than was possible with conventional VHF telemetry. This increase in the number of fixes per animal allowed the intra-home range movement patterns of animal to be examined in detail. Chapter 4 examines the movement patterns of feral cats whilst within their home range while chapter 5 examines the possible causes for the large unused areas within each home range.

Chapter 4

Movement patterns and foraging behaviour of introduced predators in tall forests

Chapter overview

In this chapter, I examine how feral cats move though their environment and discuss how these movements may indicate the foraging patterns they employ. Animals rarely use their home range in a random manner but rather are driven by the need to find resources such as food and shelter that may occur in parts of their range and to maximise the probability of successful breeding. The use of GPS collars to gather accurate and high volume locational data has allowed the intra-home range movements of the feral cats in Far East Gippsland to be analysed in ways not previously possible using conventional VHF radio telemetry.

I analysed the step lengths and turn angles of feral cats taken at three different intervals - 6 hourly, hourly and every 15 minutes - to determine if their movement patterns were purely random or followed a Lévy walk style of searching pattern. I found that feral cats employ a Lévy style walk pattern when sampled at both hourly and 15 minute intervals. Sampling at the coarser 6 hourly interval revealed that while the cats still employed a Lévy style walk pattern, it was less optimal than that found at the finer sampling intervals. This indicates that at the coarser sampling intervals may not adequately detect movement patterns.

Employing a Lévy walk search pattern increases the chances of feral cats encountering prey items that are sparsely distributed through the environment and maximises their potential hunting return for effort expended.

Knowing how feral cats move through their environment should allow control programs to be designed that target feral cats more efficiently. Using sparsely distributed control measures (traps or baits) over a large area increases the probability of a feral cat encountering and investigating the control measure.

Introduction

Animals rarely move through their individual ranges in a completely random manner. Their patterns of movement are driven by behaviours that have been selected to maximize the probability of breeding, to achieve greater efficiency in access to resources, particularly food, shelter and breeding places, and to gain familiarity with their range (Burt 1943). Study of the movement patterns of wildlife has recently attracted renewed interest, driven in part by new approaches to gathering and analysing locational data (Coulombe *et al.* 2006; Robley and Gormley 2010; Rodgers 2001).

Location data may be gathered either by physically sighting the animals or through the use of remote techniques such as radio telemetry. Such techniques are often laborious and, in the case of telemetry, require observers to take bearings and subsequently triangulate to obtain location fixes (Kliskey and Byrom 2001). Advances in Global Positioning System (GPS) technology have resulted in the miniaturisation of GPS receivers, allowing them to be placed on smaller and more mobile animals (Jiang *et al.* 2008; Rodgers 2001; Schultze and Feilitz 2001). Using GPS technology to collect data has resulted in the ability to collect much more data per animal than was previously possible with standard Very High Frequency (VHF) radio telemetry techniques (Rodgers 2001). This increase in both the quantity and quality of data has allowed intra-home range movement patterns to be both seen and examined in more detail than has been possible previously (Dassault *et al.* 2006; Kliskey and Byrom 2001; Robley and Gormley 2010).

Several alternative hypotheses have been put forward to describe these observed patterns of movement. These include random walks, composite Brownian walks (Benhamou 2007), Lévy walks (Viswanathan *et al.* 1996; Viswanathan *et al.* 1999) and adaptive Lévy walks (Reynolds 2009). Random walks have a uniform distribution of step lengths and turn angles while composite Brownian walks use a combination of step length drawn from two random walk distributions where one has a short mean step length and the other a longer mean step length (Benhamou 2007).

The Lévy walk is characterised as multiple short distance movements interspersed with less frequent but longer ranged movements that result in a long tailed power law distribution - $P(l_j) \sim l_j^{-\mu}$ with $1 < \mu \leq 3$, where l_j is the step length and μ is the slope of the regression line of the log/log relationship between step length and frequency of occurrence. Brownian motion describes purely random movement through the environment. A Lévy style walk is apparent if $1 \leq \mu \leq 3$ (Viswanathan *et al.* 1996) and Brownian motion becomes apparent when $\mu > 3$ (Bartumeus *et al.* 2005). Viswanathan *et al.* (1996) showed that for non-destructive foraging, a $\mu = 2$ resulted in an optimal search pattern for sparsely distributed prey items. Destructive foraging occurs when the entirety of a resource is consumed at a particular patch and that patch is then no longer available for foraging. With non-destructive foraging, the resource is only partially consumed at each location and the animal may return at a later stage to consume more of the resource.

Lévy walk foraging patterns are found in many species in nature including micro-organisms (Levandowsky *et al.* 1988), honey bees and moths (Reynolds *et al.* 2007a; Reynolds *et al.* 2007b), marine animals (Sims *et al.* 2008), jackals (Atkinson *et al.* 2002), elephants (Dai *et al.* 2007), spider monkeys (Ramos-Fernandez *et al.* 2004) and human hunter-gatherers (Brown *et al.* 2007). Many of the studies reporting Lévy walks involved direct observation of the subject by the researcher (e.g. Atkinson *et al.* 2002; Dai *et al.* 2007; Ramos-Fernandez *et al.* 2004). For larger or conspicuous vertebrates where sufficient distance can be kept between the researcher and the subject, this method may be employed without the presence of the observer biasing animals' behaviour (Dai *et al.* 2007). In the case of cryptic animals or in environments where sight distance is reduced, alternative methods including GPS tags or data loggers need to be used. Until recently the battery requirements of GPS collars have precluded their long-term use on mid or small sized vertebrates (Haines *et al.* 2006; Kliskey and Byrom 2001; Rodgers 2001).

Knowledge of how an animal moves through its environment is of particular relevance in the study of invasive animals that are operating in novel environments and which, in the case of introduced predators, can cause devastating declines in naive native prey species (Burbidge and Manly 2002; Burbidge and McKenzie 1989).

Knowledge of their foraging behaviour and the strategies that such species employ to improve efficiency in prey encounter, detection and capture is likely to underpin the success of any control program. For example, baiting patterns could be adjusted to suit the foraging pattern of an invasive predator and result in an increased probability of the predator finding the bait.

Fine scale data on the movement patterns of small and mid-sized introduced predators in environments where observation is impractical or introduces a bias have until now not been available. Improvements in GPS technology now allow fine scale data for such invasive predators to be obtained without biasing the animals' behaviour (Kliskey and Byrom 2001).

In this chapter I use the introduced feral cat (*Felis catus*) in Australia as a model to demonstrate the utility of modern approaches to gathering and analysing movement data and for identifying avenues for improved control. Analyses of the movement patterns of *F. catus* indicate that these introduced predators may not use simple Brownian motion in searching for prey items, but appear instead to utilise an optimal searching strategy for prey items that are themselves likely to be patchily distributed.

Methods

Study area

The approximately 10 000 km^2 Southern Ark Project area is located in far eastern Victoria in south-eastern Australia (37.34 S, 149.09 E). Most of the area comprises steep forested hills with small areas of cleared private land that are used for agriculture. Mean annual rainfall is 970 mm (see Chapter 1 for a full study site description). The feral cats used in this study were located in the West Cann UMA and the Drummer PMA. The data used for this project were gathered between February and December 2007.

Feral cat capture

Feral cats were trapped using #1.5 and #3 Victor Soft Catch® (Woodstream Corporation, Lititz, Pennsylvania; current manufacturer: Oneida Victor, Inc., Euclid, Ohio) rubber jaw traps. Trap sets were baited with either meat (chicken or beef) or 'Pongo', an olfactory lure consisting of a blend of cat faeces, cat urine and water (Algar *et al.* 2002). Half of the trap sets had a Felid Attractant Phonics (FAPs) audio lure (Westcare Industries, Nedlands, Western Australia) placed adjacent to the trap as an additional attractant. All traps were checked daily at dawn. Captured feral cats were sedated using an intramuscular injection of either Zoletil® (5.0 mg/kg) or Domitor® (0.15 mg/kg) or in combination (0.05 mg/kg Domitor® plus 3.0 mg/kg Zoletil®) depending on the level of sedation required.

GPS – VHF tag design

Sedated cats were sexed and weighed and checked for trap injuries. A 135 g GPS collar with mortality sensor (SirTrack, Havelock North, New Zealand) was fitted to those cats weighing over 2.7 kg. In order that the collar weight be less than 5% of the body weight of the cat, GPS collars were not fitted to cats weighing under 2.7 kg. The GPS collars were tasked to take fixes at either a combination of 15 minute intervals and hourly intervals or at 6 hour intervals. Battery lives at these tasking rates were about 2.5 and 5.5 months respectively. Occasionally the GPS was unable to take fixes at the tasked times, possibly due to topography, weather conditions, antennae position or a combination thereof (Dassault *et al.* 2001; Hulbert and French 2001; Jiang *et al.* 2008). Prior to fitting to the cats, the almanac on the collars was updated according to the manufacturer's instructions. I also determined the exact VHF frequency that the collars were transmitting on and entered this into an R 1000 radio receiver (Communication Specialists, Orange, California). This enabled the cats to be located using standard VHF radio telemetry techniques.

Following processing of the cats and the fitting of collars, cats that had been sedated using Domitor® were administered the antagonist Anti-sedan® (0.3mg/kg) to facilitate recovery from sedation. All cats were monitored during the recovery phase and then released at the point of capture when assessed to be free from the effects of the sedation.

The released cats were monitored initially using VHF telemetry. At the end of 5½ months when the GPS batteries were expected to be fading, attempts were then made to recapture the cats over the following 18 months to recover the GPS collars. Collars were also recovered from feral cats that had died. The data were downloaded from the collars using proprietary software from SirTrack. Data obtained from the collars were transformed from geographical co-ordinates to projected coordinated (UTM) using the DNRGarmin GPS Application (Ver 5.03.0002) (Department of Natural Resources, Minnesota) computer program. The downloaded GPS data contained an estimate of Horizontal Dissolution of Precision (HDOP) for each fix (scale 1 – 100). A small HDOP figure indicates that the satellites used to generate the fix are widely dispersed across the sky whereas a large HDOP indicates the satellites are closely grouped resulting in a loss of precision for that particular fix. Fixes with a HDOP > 4 were removed from the analyses because of the potentially lower level of precision achieved for these fixes (Moseby *et al.* 2009b).

Data for the cats with a 6 hourly GPS sampling interval were combined to provide larger data sets. Modelling of Lévy walk simulations show that using a larger dataset minimises potential bias in calculations (Sims *et al.* 2007). Step length and turn angle between each fix were determined using Hawth's Analysis Tools for ArcGIS (Beyer 2004) in ArcGIS 9.2 (ESRI Corp Redlands, California). Turn angle statistics were calculated in ArcView 3.2 (ESRI Corp Redlands, California) using the Animal Movements extension for ArcView 3.2 (Hooge and Eichenlaub 2000). Data points where the time since the previous fix was outside the parameters of 15 mins ± 1 min, 1 hour ± 1 min or 6 hours ± 1 min were discarded from the analysis. The remaining step-length data were combined for all cats. For analysis purposes, data were arranged into histograms with 200 metre intervals for the 6 and 1 hour sampling intervals and 100 metre intervals for the 15 minute sampling intervals. The frequency of occurrence at each interval was used in the analyses. Final data analyses were undertaken in Statistica 7 (StatSoft Inc 2004). Directional persistence in turn angles was determined using Rayleigh's test for uniformity (Zar 1999).

Results

Eleven feral cats were captured between January and October 2007 and fitted with GPS collars. Five collars were recovered with useable data. One collar was recovered that had failed soon after being fixed to the cat resulting in no useable data being recorded. Five collars were unable to be recovered. Between February and December 2007 the recovered GPS collars returned 2 243 successful fixes. After discarding those where the time between fixes was outside *a priori* parameters, a total of 699 fixes were obtained at 6 hourly intervals, 326 fixes at hourly intervals and 375 fixes at 15 minute intervals. The distances travelled (mean ± se) by the cats between successive fixes were 821 ± 26 m at 6 hour intervals, 185 ± 11 m at hourly intervals and 76 ± 6 m at 15 minute intervals (**Table 10**). At 15 minute and 1 hour sampling intervals there was no propensity for directional persistence between steps with turn angles evenly distributed ($z = 1.97$, $n = 375$, $p > 0.1$ and $z = 1.96$, $n = 326$, $p > 0.1$, respectively). At 6 hourly sampling intervals, the distribution of turn angles was not distributed normally ($z = 5.07$, $n = 699$, $p < 0.01$). The mean turn angle (mean ± 95% CI) at 6 hourly steps was 191 ± 14^o (Figure 11).

The distribution of the step lengths at each temporal scale showed a log power-tailed distribution (Figure 12). Regression of the $\log_{10}$ relationship between step length and frequency showed the presence of a Lévy walk pattern - $P(l_j) \sim l_j^{-\mu}$ - at each temporal scale (Figure 12).

Movement patterns at both hourly and 15 minute intervals indicated optimal searching patterns by feral cats - $\mu = 2.02$ and $\mu = 2.09$ respectively. At the 6 hourly sampling intervals a Lévy walk pattern was still apparent in the movement patterns of the feral cats but was less obvious than at the other sampling intervals. This indicates that this sampling period may be too great to detect searching patterns effectively (**Table 10**).

Table 10. Mean step length and Lévy parameter for feral cats at each sampling interval.

Interval	Mean step length (m ± se)	$r^{2\,a}$	μ [b]
6 hours	821 ± 26	0.64	1.49
1 hour	185 ± 11	0.93	2.02
15 minute	76 ± 6	0.90	2.09

[a] r^2 is the coefficient of determination for the linear regression of the log/log relationship between step length and frequency of occurrence of the step length.
[b] μ is the slope of the line for the linear regression of the log/log relationship between step length and the frequency of occurrence of the step length.

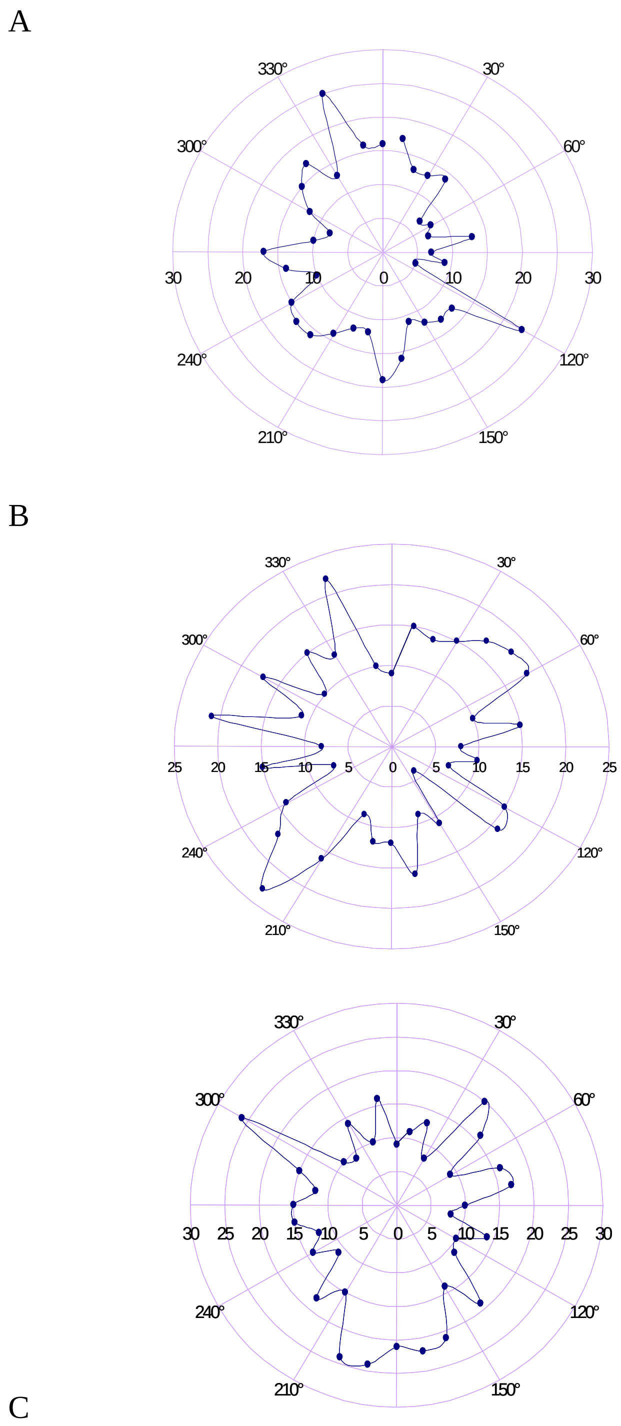


Figure 11. Distribution of turn angles for feral cats at the sampled intervals. Turn angles are grouped in 10° intervals. 0/360° indicates that the animal continued in a straight line whereas 180° indicates that the animal turned and retraced its previous track. A = 15 min sampling, B = hourly sampling C = 6 hour sampling.

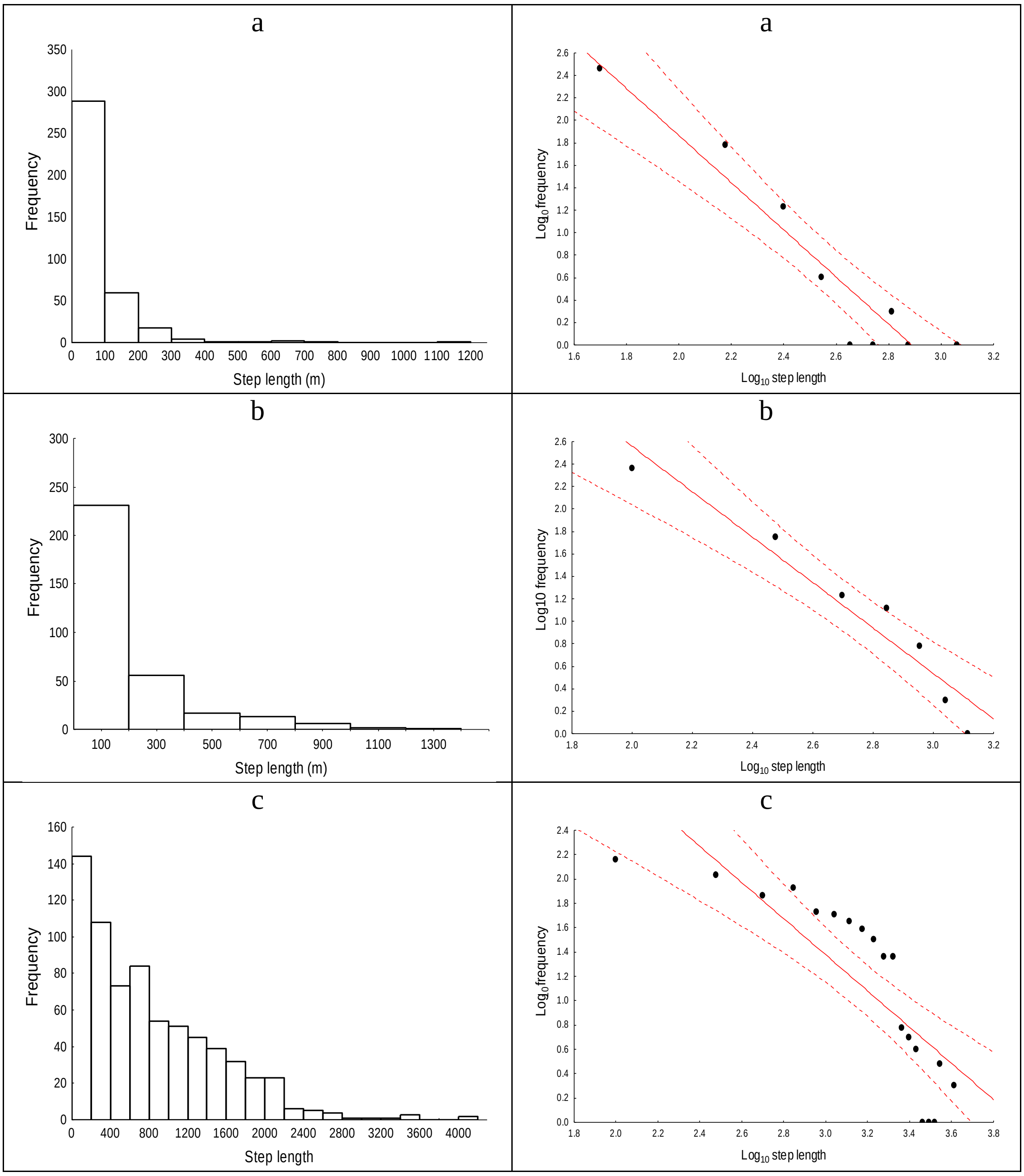


Figure 12. Distribution of step lengths for each temporal scale with corresponding log-log regression of frequency against step length. Solid line is the fitted regression line. Dotted lines are the 95% CI of the fitted line. A= 15 min sampling, B = hourly sampling, c = 6 hourly sampling.

Discussion

I analysed the movement patterns of the introduced predator, *Felis catus*, with data collected at three time scales. A normal distribution of the step lengths and $\mu \geq 3$ that would be expected from simple Brownian motion are not apparent in the data. The long tailed power-law distribution of the data and $1 < \mu \leq 3$ indicates the presence of a

Lévy walk pattern. As $\mu \approx 2$ at the 15 minute and 1 hour sampling regime, it appears that feral cats employ an efficient movement pattern while searching for resources at these temporal scales.

The Lévy walk pattern, while still present, is less apparent at the coarser 6 hourly sampling interval. This may have resulted from the sampling being too coarse to effectively detect such a pattern. If the movement pattern reflects foraging, it is unlikely that feral cats would forage for periods of 6 hours: most foraging could be expected to take place at much shorter temporal intervals. It is also likely that portions of the movement behaviour incorporated into the sampling intervals include non-foraging behaviour such as moving from and to den locations, resting, predation avoidance, mating and defence of territory. However, even at this coarse sampling interval, it is apparent that feral cats do not move through their environment in a random or Brownian manner but rather they do so with some evidently distinct purpose and pattern.

Implications for prey species

The employment of Lévy style movement behaviour by predators optimises their chance of encountering prey items when the prey are sparsely distributed across the landscape (Humphries *et al.* 2010; Viswanathan *et al.* 1996), while a Brownian (random) motion style searching pattern is more suited when prey are more abundant (Humphries *et al.* 2010). Invasive predators, such as feral cats, forage for a combination of mammalian, avian and reptilian prey items (Dickman 1996; Jones and Coman 1981; Triggs *et al.* 1984) which would not be distributed uniformly through the environment. The Lévy walk style of searching is also more advantageous if the predator is larger and faster than the prey item, while Brownian style searches are more likely to increase encounter success when the target is larger and faster than the predator (Viswanathan *et al.* 2002). Feral cats prey on animals that are on the lower end of the Critical Weight Range (35- 5500g) (Burbidge and McKenzie 1989) and, in far eastern Victoria, generally take prey weighing less than themselves (Triggs *et al.* 1984; and also see Appendix 1).

Non-destructive foraging occurs when a predator leaves an area prior to the prey items being completely extirpated, either through the predator being satiated or the prey becoming temporarily depleted. Temporary depletion can occur through a behavioural response of prey that reduced their chance of being eaten (Charnov *et al.* 1976). Optimal foraging theory (Charnov 1976; Pyke 1984) suggests that once prey behaviour or abundance is depressed to a point where continued searching by the predator is no longer viable, the predator moves to a new patch. Following the departure of the predator, prey behaviour or abundance is released (Charnov *et al.* 1976). Employment of a Lévy walk, rather than a Brownian movement style of foraging by predators, increases the time intervals between re-searching the same patch (Viswanathan *et al.* 2000; Viswanathan *et al.* 2002), which in turn should increase the time since release from depression and thus make the prey more readily available to the predator on its return to that patch.

For a naïve prey species, especially one that did not co-evolve with the predator, this depression may arise through depletion of the population rather than through individual behavioural responses (Charnov *et al.* 1976). The prey item does not recognise the danger the predator poses and therefore undertakes minimal or no avoidance behaviour (Banks 1998; Dickman 1992; Russell and Banks 2007), or avoidance behaviour that is not appropriate for the predator (Banks and Dickman 2007). As a result, the predator can forage for longer and take more prey items in the same patch before continued searching of that patch becomes energetically unprofitable (Charnov 1976; Pyke 1984). This was shown experimentally by Dickman (1992) who introduced mice from both predator-free and predator-present areas to a new area containing predators. Predator-naïve mice were depredated at a rate 2 ½ times that of mice that had been introduced from areas where predators were present.

If prey species are able to reproduce at a higher rate than they are being harvested by the predator, an increase in predation may have no real impact on the prey species population. However, if the harvest rate of the predator exceeds that at which the prey population can replace itself, then the population of the prey species must decline (Hone 1999b; Hone 2007). For a rare or endangered prey population with low reproductive capacity, an increase in predation through introduction of a novel predator may lead to extinction (Smith and Quin 1996). This has been evident on predator-free islands where cats were introduced and has resulted in the extirpation of several prey species (Burbidge and Manly 2002; Galbreath and Brown 2004; Short and Smith 1994; Vazquez-Dominquez *et al.* 2004). Similarly when populations of more common prey species erupt (for example a mouse plague or rabbits in the arid zone following a rainfall event) the increase can lead to increased feral cat populations being sustained during the period of the eruption and result in heavy *per capita* predation pressure, or hyperpredation (Smith and Quin 1996). As populations of prey species decline post-eruption, feral cats will switch to less common prey species to survive. If a secondary prey species has a low rate of increase, occurs as remnant populations or is restricted to scattered refuge sites, this can then lead to marked population declines or local extinction (Smith and Quin 1996)

Implications for control

Brownian searching techniques provide an efficient searching pattern where prey items are in high densities or are larger and faster than the predator while Lévy walk searching maximises encounter rates where prey items are sparsely distributed through the environment (Viswanathan *et al.* 2002). Modelling the distribution of control measures to suit the foraging style of the species to be controlled should increase the probability of encounter by that species with a control device. Animals employing a Lévy walk style of foraging pattern are more likely to encounter a control device (for example a trap or bait) than those that employ a Brownian foraging pattern if the control measures are sparsely placed through the environment. Trapping is the predominant technique used to control feral cats in the eastern Australian states, and the effort required to place and check traps at regular, closely spaced intervals is high (Reddiex *et al.* 2006).

As the feral cats studied here move through the forest environment using a Lévy walk, traps or control measures do not need to be closely spaced to be effective. An example of this is the aerial baiting for feral cats in Western Australia. Baits are delivered at a rate of 50 baits /km^2. For a uniform coverage, a single bait would have to be placed in every 140 x 140 m block within the baited area. However, baits are released from the aircraft in a single block of 50 per km^2 (Johnston *et al.* 2010b) (resulting in a 300 x 100 spread) or in blocks of 5 at more regular intervals (Johnston *et al.* 2011). This results in a clumped distribution of baits and, as cats employ a Lévy walk movement pattern, increases the likelihood of cats encountering the baits.

Chapter 5

Intra-home range use by feral cats

Chapter overview

Analysis of the home ranges of feral cats in Chapter 3 revealed that for each cat fitted with a GPS collar, there were large areas within the home range that they did not enter. In this chapter, I examine the potential reasons why those areas were not used by feral cats.

I hypothesised that these ‘holes’ in the ranges were areas where prey species were scarce and that, as a result, cats avoided these areas because foraging would be energetically unprofitable. To test this hypothesis, I established trapping grids to capture small prey-sized mammals within the holes. These grids bordered areas of high cat use and extended into the areas not used by cats. Analyses of the data revealed no differences in the capture rates of prey species between the unused areas and areas of high cat use, thus allowing the hypothesis to be rejected.

In contrast, modelling abiotic and vegetation community-based parameters explained much of the variation in the data. The global model was overdispersed ($\hat{c} = 6.3$) indicating that other un-measured parameters were also influencing home range use by cats. It is most likely that this unaccounted for parameter is avoidance of intraguild predation.

The collar deployed on one of the cats was recovered following activation of its inbuilt mortality sensor. This collar bore the teeth marks of a larger predator such as a dog or fox. This suggested that at least one missing parameter in the models is the presence of other intraguild predators known to occur in the study area. Cats are smaller than foxes and dogs and are likely to be subordinate to them; hence it is possible that the home range ‘gaps’ reflect avoidance by the cats of the other predators. Examination of GPS data used to determine the home ranges of feral cats in areas with no other mammalian predators failed to uncover any similarly unused areas within those home ranges, thus providing comparative support for this possibility.

The presence of higher order intraguild predators within the system inhabited by the cats may impose on them a level of behavioural suppression and confine the cats to safer parts of the forests. Reducing the density these intraguild predators may trigger a mesopredator release effect, with cats then becoming the primary predator.

Introduction

Many animals exhibit fidelity to a particular area and are therefore described as having a home range (Börger *et al.* 2008; Munger 1984). The most widely accepted definition of a home range within the literature is that described by Burt (1943) as:

> ...that area traversed by the individual in its normal activities of food gathering, mating, and caring for young. Occasional sallies outside the area, perhaps exploratory in nature should not be considered as part of the home range.

Until recently, data for determining the home range sizes of mammals have been gathered largely through direct observations of the target individuals, capturing and repeatedly recapturing animals in different locations, or through the use of VHF telemetry (White and Garrott 1990). These techniques, while useful, are laborious to carry out and usually yield relatively small data sets. The more recent use of Global Positioning System (GPS) collars to record location fixes of animals has increased the ability of researchers to look inside the home range and examine both how and why individuals use parts of their home range (Coelho *et al.* 2007; Kliskey and Byrom 2001). Very few species use or traverse their home ranges in a homogeneous manner. How an animal moves around in its home range is governed by a number of factors including the needs to find adequate food and shelter resources (Kernohan *et al.* 1998), to secure or defend mates (Kernohan *et al.* 2001) and avoid predators (Heithaus 2001). The presence of competitors or larger predators can also influence how animals utilise both the space and resources within their home ranges (Basset 1995; Harris *et al.* 1990).

Knowledge of how and why animals move through their home ranges can assist in management. Additionally, analysis of large amounts of accurate GPS data can identify the location and level of use of obligate resources, allowing the level of importance of these resources to the study animal to be determined (Lele and Keim 2006; Moen *et al.* 1996; Osborne *et al.* 2001). This type of analysis can help to determine when, where and how to focus management against pest species (see chapter 4).

Analysis of the data from the GPS-collared cats revealed that each cat had areas within their ranges that were not used and they showed a propensity to move along or near creek lines and other areas of heavy cover. I hypothesised initially that the reason why cats used some areas within their home range preferentially and avoided others was due to spatial differences in prey availability. This hypothesis was tested by live trapping small mammals that form a large part of the diet of feral cats in areas used heavily by cats and in areas that were not used. I also hypothesised that biotic and abiotic factors could influence the use of space by cats, with individuals perhaps selecting landscape attributes such as vegetation type, elevation, distance from creek-lines, and slope of the land. This hypothesis was tested using AIC based modelling. Other factors, such as the presence of large intraguild predators in the areas unused by cats, could not be tested directly, but are discussed and evaluated with reference to other works.

Methods

Study area

This project was undertaken in the West Cann Unpoisoned Monitoring Area. The holes in the home range of one cat, Olof, were selected primarily for reasons of safety. Trapping for small mammals commenced February 2009. At this time the large bushfires in Victoria (known in the popular media as the “Black Saturday Fires”) had not long been contained and were still burning in many places. Most of the field and fire staff from Parks Victoria and DSE in Cann River had been deployed elsewhere in the state. Due to the extreme fire danger rating and high winds and daytime temperatures, it was decided in conjunction with local Department of Sustainability and Environment and Parks Victoria managers that it would be unsafe to trap any deeper into the forest than the areas allowed for within Olof’s range. Accordingly, two trapping grids were laid out adjacent to creek lines commonly used by the cat but extending into the areas that were not used (Figure 13). The GPS data from the other cats fitted with GPS collars also showed that they used creek-lines more heavily than other habitats. While the grids were restricted to within the home range of one cat, they were chosen as they were representative of the habitats that were generally avoided and those that were used heavily by all the cats. Trapping concluded at the beginning of December 2009.

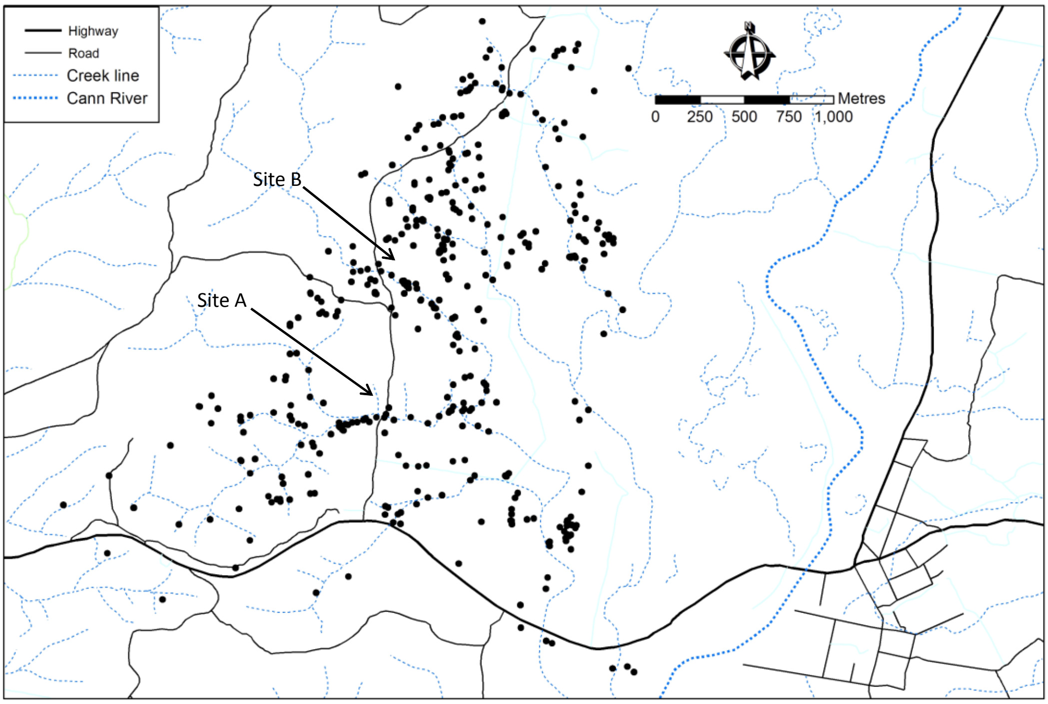


Figure 13. Location of trap grids within the home range of a feral cat, Olof in Far East Gippsland. Grid sites (A and B) are marked with arrows and the GPS locations for Olof by dots. The township of Cann River is located in the bottom right of the map. Background GIS layers were supplied by the Department of Sustainability and Environment, Victoria.

Small mammal trapping

The two trapping grids were established within the home range of the feral cat Olof. The grids had one side adjacent to areas of high use as shown by the GPS data and extended from there into the unused areas within the home range. Each grid was trapped for three consecutive nights on three occasions. Each grid contained 64 type A Elliott traps (330 x 100 x 100 mm) (Elliott Scientific Equipment, Upwey, Victoria) in an 8 x 8 grid with 20 m spacing between the traps. Traps were placed in the most suitable location within 2 metres of the 20 metre point to maximise captures (Stewart 1979; Tasker and Dickman 2002). A wad of dacron fibre was placed in each trap to provide bedding and warmth for captured animals. Each trap was also placed in a plastic bag to prevent the ingress of water and dew. Traps were positioned in such a way to as to be shaded from the sun and, if necessary, additional bark and leaf litter were added to ensure that traps remained shaded and to camouflage traps from casual observance. Traps were baited with a mixture of peanut butter, rolled oats and honey (Catling and Burt 1994; Dickman *et al.* 1983) compressed into a 2 cm ball. To reduce potential bias from differing proportions of ingredients, the bait mixture was made to the same "recipe" on each occasion (see chapter 3 for details).

Traps were checked from dawn each day with captured animals identified, sexed and weighed. Each animal was fitted with an individually numbered lightweight aluminium (< 0.18 g) ear tag (Model 1005-1, National Band and Tag Co, Newport, Kentucky). The ear tags are self-piercing and were fitted using the proprietary applicator. To prevent infection, the ears were swabbed with Betadine® solution pre and post fitting of the tags (Sharp *et al.* 2007). Following processing the animals were released at the point of capture and observed until they entered suitable shelter.

GIS data

The methods used for capturing feral cats and details of the GPS collars are discussed in Chapter 3. The GPS fixes for each cat for which GPS data were available were entered into ArcMap 9.2 (ESRI Inc. Redlands, California). Each point was buffered with a radius of the mean estimate of accuracy of all points (in my case 18.54 m). This was calculated by averaging all horizontal dissolution of precision (HDOP) values and multiplying that by the receiver accuracy (6 m) (Moseby *et al.* 2009b). Polygons were created outlining the unused areas in the home ranges but excluding the buffered areas around each point. Random points were created within polygons using Hawth's Analysis Tools for ArcGIS (Beyer 2004). The same numbers of random points were created as there were useable fixes for each cat. To determine if the cat preferentially used areas within its home range, the parameters of each actual fix were compared against the randomly generated locations in areas unused by the cat. Abiotic environmental parameters thought to influence the location of the feral cat were determined *a priori.* These values were determined for each point by interrogating the appropriate GIS layers using the intersect point tool in Hawth's Analysis Tools for ArcGIS (Beyer 2004). The parameters selected to be examined were:

1. Distance to nearest stream (metres)
2. Elevation (metres above mean sea level)
3. EVC code (This is a categorical factor where each vegetation community is classified using a code. See Davies *et al.* (2002) and DSE (2010b) for additional details if required)
4. Slope of the land at the point
5. Aspect of the land at the point, and
6. Distance to nearest roadway (metres).

Analysis

The minimum numbers of animals known to be alive (MNKA) (Gilbert and Krebs 1981) were calculated for each potential prey species and for prey animals as a whole. Prey density, population estimates and detection probabilities were calculated for each trapping grid using DENSITY software, version 4.4 (Efford 2004). The mean distance moved by animals between captures was calculated and added as a buffer to the grid size to remove potential bias in prey density estimations by accounting for the area used by animals captured on the edges of the grids. Prey species were combined for this analysis to give overall prey density rather than individual species density. Chi-squared analysis was used to identify differences in capture rates between trap lines close to the creek lines and those further away. As cats did not appear to use roads, chi-squared analyses were used to determine if distance from road affected capture rates.

A stepwise logistic regression with binomial distribution was used to determine if the abiotic environmental parameters determined *a priori* were suitable parameters for inclusion in the AIC models. This analysis was undertaken for each cat and for the dataset as a whole and compared the parameter values for each known location of a feral cat against those of the randomly chosen locations in areas unused by the cats.

All analyses detailed in this section were undertaken using the Statistica 7 statistics package (StatSoft Inc 2004).

Modelling

Once the appropriate abiotic factors were determined using the logistic regression model, ten *a priori* models, including a global model which contained all the parameters, were constructed and tested. The models were used to determine which would best fit the home range use by feral cats based on the abiotic environmental factors. The performance of the models against the GPS data were compared using Akaike Information Criteria (AIC) (Burnham and Anderson 2002). It is suggested that when the sample size to parameter ratio (n/K) is < 40 that AIC be corrected to AIC_c (Burnham and Anderson 2002). As the ratio of sample size to parameters in this analysis was about 118, this correction was not used. The overdisperson parameter ($\hat{c}$) of the models was calculated from the goodness-of-fit chi-square statistics of the global model (i.e. all parameters included). This is divided by the degrees of freedom of that model to give a $\hat{c}$ value. AIC analyses were undertaken using the GLZ model building function of the Statistica 7 statistical package (StatSoft Inc 2004).

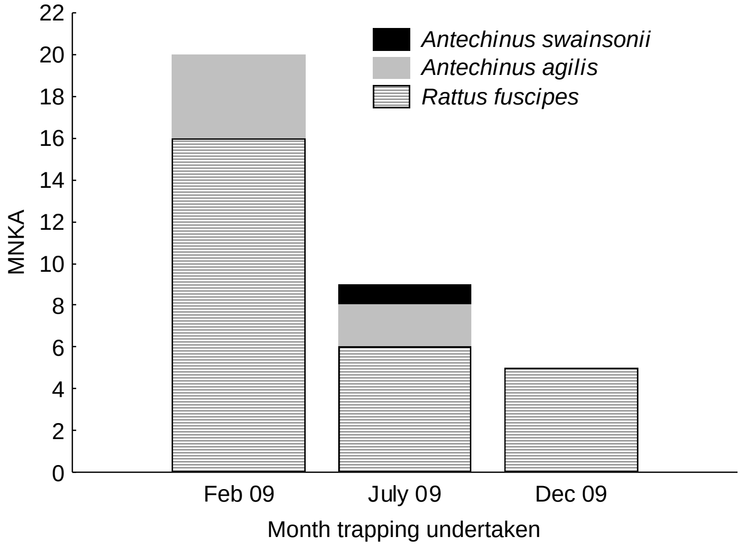
Results

Small mammal trapping was undertaken for 1 152 trap nights over three occasions commencing in February 2009 and concluding at the beginning of December 2009. Overall, 37 unique animals were captured a total of 73 times. Three species of small mammal were captured – bush rat (*Rattus fuscipes*), agile antechinus (*Antechinus agilis*) and dusky antechinus (*A. swainsonii*) with captures dominated by the bush rat. Capture and recapture rates at site B were quite low, with correspondingly large standard errors for population size and density estimates. Only a single animal was captured in February 2009 resulting in no estimation of density or capture probability (Table 11). The estimated overall population size declined at site A during the course of the trapping but increased at site B over the same period (Figure 14). Insufficient animals were recaptured between trapping sessions to allow a population estimation to be made over the term of the sampling.

Table 11. Population and capture-related estimates for small mammals at each of two trapping sites (A and B) in Far East Gippsland. As all species captured are potential prey for feral cats, numbers have been combined to yield overall estimates of prey abundance. Note: only a single animal was captured during the February 2009 trapping session at site B so no density or capture rate estimates are available. MNKA = minimum number known to be alive.

Site and date trapped	MNKA	Recaptures	Population estimate ± se	Density ± se (animals ha^{-1})	Capture Prob
Site A					
February 2009	20	16	21 ± 1.65	5.48 ± 1.32	0.57
July 2009	9	7	9 ± 0.98	2.53 ± 0.93	0.59
December 2009	5	2	6 ± 2.02	2.27 ± 1.55	0.38
Mean			12 ± 4.58	3.43 ± 1.03	0.51 ± 0.07
Site B					
February 2009	1	0	1 ± 1.07	*	*
July 2009	2	3	2 ± 0.1	0.42 ± .023	0.83
December 2009	7	1	17 ± 12.42	3.90 ± 3.54	0.16
Mean			7 ± 5.17	1.97 ± 1.92	0.49 ± 0.33

Data from site B were pooled into grid halves and analysed as the half closest to the creek-line or road and the half furthest from the creek-line or road to ensure sufficient numbers of captures. Distance from the creek-line had no effect on numbers of captures, with no differences found in the number of animals captured on each line of the grid at either trapping site (Site A: = 8.39, *P* = ns and Site B: = 0, *P* = ns). Distance to the road also had no effect on the capture rate, with no difference found between the number of captures on lines closest to the road compared with those further away (site A: = 7.31, *P* = ns and site B: = 2.57, *P* = ns).



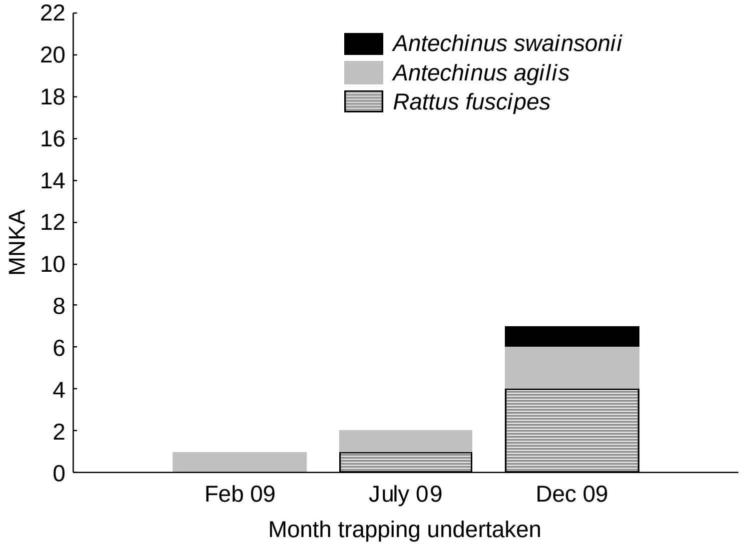


Figure 14. Minimum numbers of small mammals known to be alive during three trapping sessions in Far East Gippsland. MNKA = minimum number known to be alive, a = site A, b = site B.

GPS data were obtained and analysed for four feral cats. A total of 768 useable fixes were gained from the four cats. I did not use the fixes taken at 15 minute intervals for the cat Hans as on many occasions the subsequent fix location was still within the GPS error radius of the previous fix. The same numbers of random fixes were generated, making a total of 1 536 data points used in the analyses. Slight variation in the time between fixes ensured that fixes were taken at all times during the day over the period the collars were on the cats (see Chapter 3 for a full explanation).

The GPS collar of one cat, Olof, was found after the mortality sensor was triggered. The cat had been radio tracked two weeks previously. It was assumed to be alive at that time as it had moved between successive fixes and the mortality sensor had not been triggered. The collar was located but not the body of the cat. The epoxy resin used to protect the components of the collar was damaged. An examination indicated that the damage was caused by the teeth of a large predator; either a fox or wild dog as shown by the spacing of the teeth marks. It is unknown if the cat died as a result of predation or had died and was subsequently consumed as carrion.

Logistic regression of the biotic and abiotic environmental parameters showed that their relevance was specific to each individual cat, with some parameters being considered significant for some cats but not others. From evaluated parameters, five were selected as they had a significant effect on the presence of most feral cats, these being elevation, slope, aspect, ecological vegetation classification code (EVC), and distance to stream (Table 12). A sixth factor, distance to road, was also added into the analyses as it had been shown to be a significant factor for several cats. Model averaging was not used as the second best model contained all factors (Table 13).

Table 12. Result of logistic regression of all factors considered *a priori* to influence feral cat home range use. Factors are shown in increasing order of significance as determined by the analysis. *EVC is a categorical variable with eight categories giving it 8 degrees of freedom within the analysis.

Factor	df	Wald statistic	*P*
Intercept	1	0.00002	0.996
Distance to road	1	0.371	0.542
Aspect	1	0.399	0.046
Elevation	1	7.229	0.007
EVC*	8	27.372	0.001
Slope	1	21.416	≤0.001
Distance to stream	1	27.776	≤0.001

Table 13. Summary of the model selection statistics for the 10 *a priori* determined models for describing cat presence at study sites in Far East Gippsland. K is the number of parameters in the model (EVC contributes 8 parameters to each model it is included in). -2Log(L) is twice the negative log likelihood value. AIC is the Akaike Information Criteria value and Δ AIC is the difference in AIC values between that model and model 1. (Model terms: Elev = elevation, Stream = distance to nearest stream, Slope, angle of the slope of the land and the sampling point, Aspect = aspect of the land at the sampling point, EVC = Ecological Vegetation Classification code (note this is a categorical factor not a continuous factor), and Road = distance to nearest road).

No.	Model	K	-2Log(L)	AIC	Δ AIC
1	Elev, Stream, Slope, EVC, Aspect	12	2047.40	2073.40	0
2	Elev, Stream, Slope, EVC, Aspect, Road	13	2047.03	2075.03	1.63
3	Elev, Stream, Slope, EVC	11	2051.49	2075.49	2.09
4	Elev, Stream, Slope	3	2083.89	2091.39	17.99
5	Stream, EVC	9	2071.62	2091.62	18.22
6	Elev, Stream, EVC	10	2071.61	2093.61	20.21
7	Stream	1	2098.39	2102.39	28.99
8	Elev, stream	2	2098.35	2104.35	30.95
9	EVC	8	2098.40	2116.40	43.00
10	Elev	1	2128.62	2132.62	59.22

The model with the lowest AIC value generally best describes the variation in the data and is used as the model of best fit. However, in this case, model 3 is chosen as the best model. As the Δ AIC is about 2 or lower for all three of the best models, there is substantial support for each of these models. Also, the additional parameters of "aspect" in model 1 and "aspect" and "distance to road" in model 2 have added no more to the model than would occur through adding a further parameter which has no effect on the model outcome (Burnham and Anderson 2002). As a result, models 1 and 2 are discarded in favour of model 3 on the basis that the added parameters in those models have added little, if any, value to the model.

The global model produced a $\hat{c} = 6.33$ indicating that the models were overdispersed. As $\hat{c} > 4$, it also indicates that the fitted model does not account for sufficient variation in the data to be robust and that one or more unknown factors are likely to be contributing to the overall variation (Burnham and Anderson 2002 pp. 68).

Discussion

The results provided little support for the two initial hypotheses of home range use by cats. There was little evidence to suggest that cats avoided parts of their home range because they lacked either food or habitat resources. Small mammal trapping indicated that there were populations of prey animals in areas that were used and not used by the cat Olof, with prey densities of up to 5.48 ha^{-1} and a mean of 2.79 ha^{-1} across the two trapping grids. This would indicate that there are sufficient food resources within these holes to make hunting energetically profitable. In addition, there was no difference in the capture rates of potential prey items near or away from the creek-lines, suggesting no reduction in prey availability with distance from areas that cats frequent. As a result, it seems unlikely that cats avoid some areas due to the lack of available prey items. Similarly there was no reduction in captures on grid lines closer to roads than on those further away, indicating that little or no edge effect exists in relation to prey availability (Andren and Angelstam 1988; Goosem 2002).

The habitat variables chosen to explain home range spatial use by feral cats were generally cat-specific, with some being important for one cat but not for others. Five of the six variables that were chosen for modelling cat presence against the random cat absence points were shown by logistic regression analysis to be significant predictors of cat locations. Of the individual variables, distance to creek-line was the most significant in determining cat locations compared with randomly generated points. This may be due to the more complex vegetation that occurs along creek-lines (pers. observation) which affords them more shelter from weather or greater protection from intraguild predation.

The first two models in the AIC analysis were discarded as the addition of more parameters did not decrease the AIC score by much more than 2. This indicates that these additional parameters contributed little, if anything, to the ability of the model to account for the variance in the data. The calculation of the AIC values from the data uses the formula $AIC = -2\log(\mathcal{L}(\theta \mid y)) + 2K$, where $(\mathcal{L}(\theta \mid y))$ is the model likelihood and K is the number of parameters. Generally, as the number of parameters increases, the first term $(-2\log(\mathcal{L}(\theta \mid y))$ decreases. However, this is counter balanced by the increase in the second term (2K) (Burnham and Anderson 2002, p. 62). Generally if the number of parameters is increased by one, and the AIC value decreases by <2, then that parameter has not added value to the model and the decrease in AIC is attributable to the addition of the parameter.

The accepted best model, model 3, was similar to the logistic regression in showing that the most important parameters are the distance to nearest stream, slope, elevation and vegetation composition. However, as the global model was overdispersed ($\hat{c}$ = 6.3) it is likely that that some other, unmeasured factor(s) contributed to the variation within the data. It is most likely that this missing parameter involves predator avoidance strategies being employed by the cats that also contribute to determining how they use their home ranges. Two other methodological possibilities exists which may account for why cats use only some parts of their home ranges and these are discussed later.

The mammalian predator guild in Far East Gippsland consists primarily of three species, the wild dog/dingo (*Canis familiaris / C. lupus dingo*), the red fox (*Vulpes vulpes*) and the feral cat. A fourth predator, the spotted-tailed quoll (*Dasyurus maculatus*) also exists in the region but its abundance is far lower than that of the other three predators. The three main predators exhibit extensive dietary overlap, with common ringtail possum (*Pseudocheirus peregrinus*), *Antechinus* spp. and *Rattus* spp. featuring prominently in the diets (Diment 2010; Triggs *et al.* 1984; Appendix 1 of this thesis). Intraguild predation occurs between cats and quolls (Glen *et al.* 2010). Inclusion of intraguild species, particularly cats, in fox diet was found in Far East Gippsland by both Diment (2010) and Triggs *et al.*(1984). Similarly, dogs in New South Wales forests adjacent to the Gippsland border have been shown to include cats in their diet (Lunney *et al.* 2001; Lunney *et al.* 1990).

Over the period of this study, 10 wild dogs / dingoes and nine foxes were captured in the same areas as the cats. The dogs were released at the point of capture and the foxes participated in other experiments being conducted in the area (i.e. Diment (2010)). Sand-plot and bait-take data collected by Southern Ark indicates that wild dogs / dingoes and foxes are present across all the monitoring areas within the

Southern Ark Project. Approximately 43% of all bait-take is by these species (Diment 2010, p. 61). It is likely that predator avoidance is why cats avoid large areas of their home ranges. Many lower order intraguild predators face a trade-off between obtaining sufficient prey to meet their nutritional requirements and their need to remain safe from being preyed upon by higher order intraguild predators (Sih 1980). Intraguild predator avoidance strategies occur in bobcats (*Lynx rufus*) when in the same area as coyotes (*Canis lutrans*) (Wilson *et al.* 2010). When prey availability is high, bobcats avoid areas that are used extensively by coyotes. Conversely, when prey abundance is low, they are forced into areas of high coyote use.

Other opportunistic predators, such as wolf spiders (*Pardosa milvina*) prefer to remain in areas where prey items can be exposed yet where sufficient protection remains for them to reduce the potential for intraguild predation (Schmidt and Rypstra 2010). Similarly, bottlenose dolphins (*Tursiops aduncus*) avoid good quality foraging areas when tiger sharks are present, even in low numbers (Heithaus and Dill 2002). In the present study, prey animal trapping suggested that sufficient prey resources existed in the areas frequented by cats during the time the cats wore GPS collars. As a result they were not compelled to enter areas that are probably more frequently used by foxes and dogs, such as roadways and ridgelines (Mahon *et al.* 1998; Webbon *et al.* 2004) that would expose them to a higher risk of depredation.

Other studies have been conducted using GPS collars on feral cats. Hilmer (2010) and Johnston *et al.* (2010b) tracked 15 feral cats on Dirk Hartog Island, Western Australia. Feral cats were also tracked using GPS on French Island, Victoria (M.J. Johnston, unpublished data). Intraguild predators do not exist on either of these islands. The holes noticed in the home ranges of cats in this study were not present in those of cats on the islands without other predators.

If the predator teeth marks on the collar of the cat Olof resulted from an intraguild predation event, it is possible that it was forced to forage in areas frequented by foxes and wild dogs. There was a downward trend over the period of this study in the number of prey items detected on the grids within its home range. If it became energetically unprofitable to continue to forage in areas that afforded greater protection from encountering intraguild predators, it may have been forced into areas where there was an increased likelihood of encountering foxes or dogs. Unfortunately the GPS collar on Olof ceased to function several months prior to his death so this hypothesis cannot be tested.

Two methodological possibilities exist that may also explain the avoidance of these areas by feral cats. Firstly, if GPS fixes were not taken at times when cats were in the ‘unused’ areas, their avoidance of these areas would be an artefact rather than reflecting reality. The mean time between the GPS taking a location was slightly shorter than requested (see Chapter 3, Table 5), resulting in a “fix creep”. For example, using the collars with a 6 hourly sampling rate, the mean time between fixes was 5.96 h, resulting in a cumulative time slip of 10 minutes per day in the timing of fixes. A collar that took a fix at 6 am one day would take the same scheduled fix at 5.50 am the following day and at 5.40 am the next day, and so on. However, such fix creep would be very unlikely to result in apparently unused areas within home ranges. If feral cats used certain areas at certain times of the day, this would have been evident by the presence of data points at those times as a result of the time slip between fixes.

Secondly, it is possible that other cats were present in the areas not frequented by the GPS-collared cats, and that their presence deterred the collared cats from entering and using them. However, this seems unlikely as female cats appear to readily overlap both the home range and core areas of other females and male cat home ranges readily overlap those of females (see Chapter 3). The potential for exclusion by another cat only appears probable when both cats are male, however, male home ranges are substantially larger than the un-entered areas within the collared cat home ranges therefore making this scenario unlikely.

If the above line of thinking is correct, the presence of higher order intraguild predators in the environment inhabited by the cats may influence both their movements and the areas they use within their home ranges. Dingoes have a controlling effect on foxes and feral cats through direct predation (Edwards *et al.* 2002; Paltridge 2002). Additionally they can force behavioural changes, a response to an increased risk of predation, which can change both the habitat use and abundance of the mesopredator (Ritchie and Johnson 2009). Glen *et al.* (2007a) found evidence that dingoes can limit both lower order predators and prey species. Similarly, Molsher (1999) found that when fox abundance was reduced, feral cats began to move into areas previously unused when foxes were present in larger numbers. If the feral cats in Far East Gippsland do not use areas within their home range to minimise the potential for intraguild predation, the management of the higher order intraguild predators through population reduction may result in mesopredator release. This may allow feral cats to increase in abundance (Johnson *et al.* 2007; Letnic *et al.* 2009) and become the apex predator in the region. If so, this has the potential to have substantial negative impacts on prey species populations within the system (Fan *et al.* 2005; Risbey *et al.* 2000; Soulé *et al.* 1988).

It is recommended that a long-term, replicated predator removal study, similar to that of Risbey *et al.* (2000), be undertaken in Far East Gippsland to determine if the presence of dingoes / wild dogs and foxes is suppressing the behaviour and abundance of feral cats. Such an experiment should incorporate experimental control sites, sites where the population of the higher order predators is reduced individually and sites where both higher order predators are reduced. This will allow the level of suppression each exerts on feral cats to be determined. Having sites where the abundance of both higher order predators is reduced will determine whether their presence has a synergistic suppressive effect on feral cats. Monitoring of the impacts of feral cats on prey species would also need to be undertaken to determine if reducing the abundance of the higher of order predators impacts negatively on prey species populations.

If the populations of the higher order predators are reduced and a mesopredator release occurs, there are limited techniques available to land managers in Far East Gippsland to enact feral cat management through felinicide. No poison baits have been approved for use with feral cats in Victoria (APVMA 2011). In 2008 the Victorian government banned the use of rubber-jawed traps for capturing feral cats on public land (see Division 3 of the Victorian *Prevention of Cruelty to Animals Regulation 2008*). Indeed, Section 32 of that Regulation requires that any cat captured be taken to the nearest local council rather than be destroyed at the point of capture. As a result, the management of feral cats may well need to be undertaken through managing the populations of higher order intraguild predators rather than the cats directly. A study such as that suggested above would be able to guide the land managers on the best methods of doing such.

Chapter 6

Prey population dynamics and optimal baiting times for feral cats

Chapter overview

As feral cats are obligate carnivores, knowledge of the population demographics of their prey species is essential for determining the optimal timing of effective management programs. Feral cats generally prefer live prey and do not scavenge except during times of high food stress. In the eastern states of Australia such times generally occur in winter and early spring when their primary prey sources are naturally at their lowest levels, particularly after the breeding period of *Antechinus* species, which are among the most common and abundant potential prey for cats. Antechinuses are semelparous marsupials in which all males die after a brief mating period that lasts about two weeks during the winter – early spring.

To determine the population demographics, breeding cycles and population fluctuations in terrestrial mammals that could potentially form prey for feral cats in Far East Gippsland, I established four mammal trapping grids. Each grid was trapped for three consecutive nights on four separate occasions – three for small mammals and once for mid-sized mammals. Captured animals were tagged to allow recaptured animals to be identified.

Five species of prey-sized animals were captured, with the bush rat *Rattus fuscipes* being captured most commonly. Patterns of abundance of prey animals varied between the trapping grids, with the populations declining on half the grids and increasing on the remaining two grids as the study progressed. The spatial and temporal variation in these populations is most likely explained by a combination of the natural population fluctuations of the study species and site-specific phenological changes or availability of other resources in the forest over the period of sampling.

Female *Antechinus agilis* in September had 1 – 1.5 cm long pouch young, indicating that the breeding period for this species is early to mid August. Male *A. swainsonii* had entered the trappable population by the end of November. The time between conception and entering the trappable population is about five months, indicating that the breeding season for this latter species occurs before late June in the study area. The breeding seasons for the *Antechinus* species in this region occur earlier than would be expected based on the breeding periods of other populations of the same species. This has ramifications for managers in that the winter food deficit for feral cats occurs earlier than would be expected in other regions. Additionally, management intervention using poison baiting needs to avoid periods when non-target species are at increased risk. Timing of poison baiting campaigns in Far East Gippsland would have to occur earlier to avoid placing females with pouch young at risk as they are present in the forest earlier than would be expected in other regions.

Introduction

Feral cats (*Felis catus*) are obligate carnivores (Bradshaw *et al.* 1996; Zoran 2002). The loss of key metabolic enzymes has resulted in the cat having a narrowly defined diet that is high in protein and low in carbohydrates. For feral cats – that is, animals that do not rely on humans for any portion of their dietary intake – this can be satisfied only by a diet consisting largely of vertebrate prey (Zoran 2002). Feral cats usually have a preference for live prey and rarely scavenge carrion unless they are hungry and food stressed (Algar *et al.* 2007; Leyhausen 1979).

Generally the diet of feral cats consists predominantly of mammals with only limited numbers of birds being consumed (Jones 1989). In the Far East Gippsland region of Victoria, feral cat diet consists primarily of small to mid-sized mammals such as *Antechinus* spp., *Rattus* spp. and *Pseudocheirus peregrinus* (common ringtail possum) (Triggs *et al.* 1984; also see Appendix 1 of this thesis). However, the composition of prey species' populations themselves is largely unknown in Far East Gippsland; it is also not known if feral cats in that region consume different prey species in the same ratio at which they are available in the environment, or if they selectively depredate some species in preference to others.

Reducing feral cat populations through poison baiting relies on presenting baits at a time when they are most likely to be consumed by cats. This is usually during times of food stress when prey populations are at low levels such as during winter and early spring and following boom periods in prey availability when prey populations are declining (Algar *et al.* 2007; Letnic and Dickman 2010). In order to successfully manage cats through poison baiting it is essential to know at which time of year a prey deficit is most likely to occur.

Members of the marsupial carnivore genus *Antechinus* are entirely semelparous. All males die following a brief and synchronous breeding period in mid to late winter (Morton *et al.* 1989; Wakefield and Warneke 1967; Wood 1970). The mating period of *Antechinus agilis* (agile antechinus), a common and widespread species in Far East Gippsland, is triggered every year when a particular threshold is reached in the rate of change in photoperiod as days lengthen following the winter solstice (McAllan and Dickman 1986; Scott 1986). However, there is considerable variability in the timing of breeding between the various populations in Victoria (Dickman 1982; McAllan and Dickman 1986), perhaps as a result of undetected cryptic species or as a result of slippage in the photoperiodic response of panmictic species populations. As a result, it is not possible to predict the timing of the breeding period in Far East Gippsland as accurately as it is with more northern populations. Understanding and accurately predicting the time of breeding is important in helping to identify the period when populations decline and cats, in consequence, may face a much reduced prey base.

The timing of the breeding period of *A. swainsonii* (dusky antechinus) is thought to be based on altitude rather than latitude, and is much more difficult to predict accurately than in most other species of the genus (Dickman 1982; McAllan *et al.* 2006); its timing in Far East Gippsland is not known. As in *A. agilis*, there is a halving or more of the number of *A. swainsonii* available to be consumed as prey immediately following the breeding period due to the complete die-off of males. As adult females are often only half as large as adult males (McAllan *et al.* 2006), the biomass of available food for feral cats is reduced by considerably more than half following the male die-off.

Rattus fuscipes (bush rat) is also a common component of the diet of feral cats in Far East Gippsland (Triggs *et al.* 1984; also see Appendix 1 of this thesis). It is a member of the family Muridae and is one of the new endemic rodents (Watts and Aslin 1981).

It is an omnivorous species that can breed several times a year if there are sufficient food resources available (Menkhorst and Knight 2001; Taylor and Calaby 1988); while some research has focused on this species elsewhere in Victoria (e.g. Robinson 1988; Warneke 1971), little is known about its reproductive behaviour or population dynamics in Far East Gippsland.

Few data are available on the population parameters of prey species in Far East Gippsland that are likely to be relevant to managers charged either with the conservation of these taxa or the control of their predators. Parameters such as population size, species composition, the mass of small or mid-sized mammals, if sexual dimorphism exists, and if so, the sex ratios within the population, are all likely to be relevant to predators that need to balance energy expenditure against energy gain when pursuing food. Knowledge of population parameters is also essential for local land managers who are required to make decisions about the timing of certain management actions. For example, poison baiting at a time when non-target species or females with pouch young are most susceptible could place those females and young at high risk and also have disproportionately large effects on their populations.

To determine the most appropriate time to manage feral cats using poison baiting, I examine the composition and abundance of populations of small and mid-sized terrestrial mammal species that may be susceptible to feral cat predation in Far East Gippsland. I also compare the potential availability of different species with current knowledge of the diet of feral cats in the same region. I also describe the population demographics of three of the more common species.

Methods

Study area

Sampling was undertaken in the West Cann unpoisoned monitoring area, the Drummer poisoned monitoring area and the Coopracambra National Park in Far East Gippsland (Figure 15).

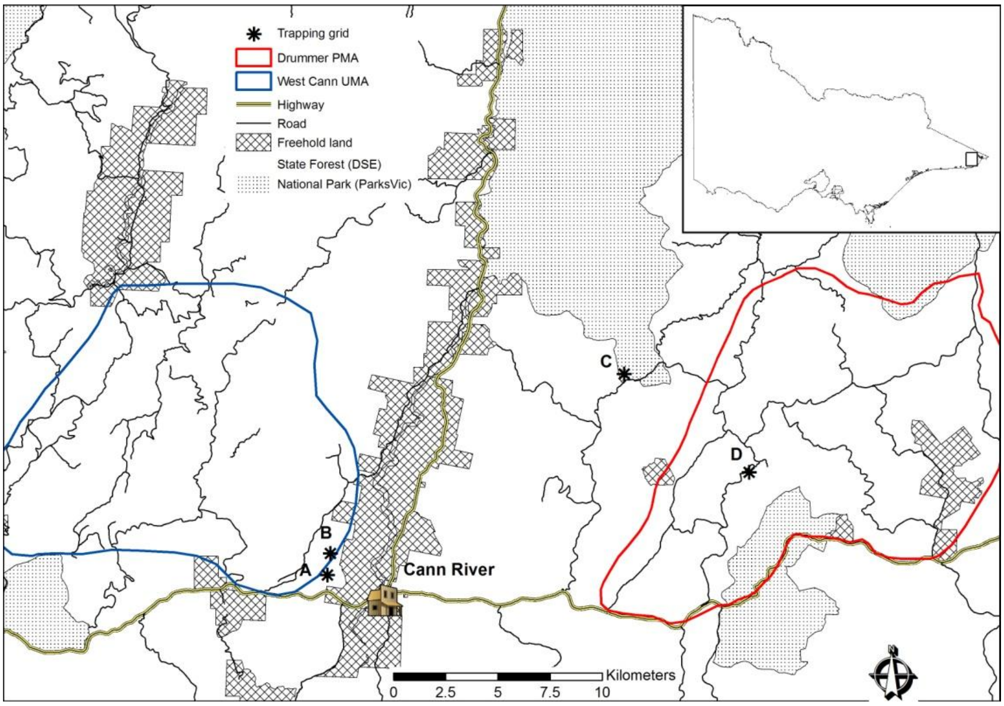


Figure 15. Location of four small and mid-sized mammal trapping grids (A – D) in Far East Gippsland, Victoria. Sites A and B are closely spaced within the home range of one feral cat (see chapter 5 for details). Sites A and B were trapped simultaneously as were sites C and D except as described in the text. Background data were extracted from geospatial data provided by the Department of Sustainability and Environment, Victoria.

Trapping

Four grids were established for trapping prey-sized mammals. The grids were placed both inside and outside the monitoring areas used by Southern Ark (Figure 15) to account for any variation in density that may occur due to differences in habitat and management regime. The grids were established in tall forest dominated by white stringy bark (*Eucalyptus globoidea*) and brown barrel (*E. fastigata*). Each grid consisted of 64 Elliott traps (330 × 100 × 100 mm) (Elliott Scientific, Upwey, Victoria) laid in an 8 × 8 grid with 20 m spacing between traps. Traps were placed at the most suitable point within 2 m of the 20 m point to maximise capture rates (Stewart 1979; Tasker and Dickman 2002). Traps were baited with a combination of peanut butter, honey and rolled oats rolled into a 2 cm ball (Catling and Burt 1994; Dickman *et al.* 1983). Bait was made to the same recipe on each occasion to reduce the potential for bias from differing proportions of ingredients (see chapter 5 for details). A wad of Dacron fibre was placed in each trap for bedding and each trap was placed in a plastic bag to prevent the ingress of rain or dew. Traps were placed in shaded positions and, where necessary, additional bark and leaf litter was added to ensure the trap remained shaded.

Wire cage traps (520 × 200 × 200 mm) (Mascot Wireworks, Homebush, New South Wales) were obtained through the Cann River Parks Victoria office for mid-sized mammal trapping. Traps were deployed in a 3 × 8 grid pattern of 24 traps at each site, and were laid out on grid lines used previously for the small mammal trapping. Cage traps were trigger-operated and closed when an animal pulled on the bait at the rear of the trap. Several large handfuls of leaf litter were added to each cage trap to provide bedding and warmth for captured animals. Each trap was fitted with a custom-made heavy PVC cover to reduce the amount of rain or dew that could affect the captured animals.

Small mammal trapping was undertaken on three occasions at sites A, B and C but occurred only twice at site D. Each site was trapped for three consecutive nights on each occasion. Mid-sized mammals were trapped on one occasion consisting of three nights of trapping at each of the four sites.

Traps were checked each day at dawn with captured animals identified, checked for sex and reproductive status, and weighed. Fresh captures were fitted with an individually numbered ear-tag made from lightweight aluminium (< 0.18 g) (Model 1005-1, National Band and Tag Co, Newport, Kentucky). These self-piercing ear-tags were fitted to the lower proximal region of the pinna using the proprietary applicator. Care was taken to avoid blood vessels when fitting the tag. The ear was swabbed with Betadine® solution (Sanofi-Aventis Pty Ltd, Rydalmere, New South Wales) both pre and post fitting of the tag to minimise infection risk (Sharp *et al.* 2007). Animals were released at the point of capture and were observed until they reached suitable shelter.

Analysis

The minimum number of animals known to be alive during each trapping session was calculated for each prey species captured. Insufficient numbers of some species were captured at some sites to allow population level analyses to be undertaken, but data for bush rats (*Rattus fuscipes*) were analysed using Density software (version 4.4)

(Efford 2004). Bush rat density was calculated based on the effective grid size rather than the actual grid size used. Effective grid size was calculated by adding a buffer zone to the grid equivalent in width to the mean distance moved by animals between captures, following conventional practice (Hayne 1949; Southern 1979). Comparisons between species and sex ratios of captures were undertaken using chi-squared tests, with Yates' correction for continuity applied in analyses with a single degree of freedom (Fowler *et al.* 1998 p. 116). Student's *t*-tests were used to determine differences in weight between males and female. These analyses were undertaken using the Statistica 7 software package (StatSoft Inc 2004).

Results

Trapping was undertaken between February 2009 and September 2010. Access to trapping grids was restricted by heavy rain or high fire danger on several occasions, resulting in the planned trapping sessions being delayed. In all, 2 046 trap nights were completed with 93 individual animals being captured on 176 occasions. One animal escaped prior to processing so has been excluded from the analyses. Five species were captured (Table 14). Overall trap success rate was 8.6%; however, trap success was high for small mammals (9.7%) compared with mid-sized mammals (1.7%). Bush rats were the most common species captured across all grids (= 47.8, *P* <0.001) (data for the three least common species were pooled for this analysis).

Table 14. Mammal species captured at four sites in Far East Gippsland, indicating total numbers of individuals captured, with recaptures shown in parentheses. Numbers of individual males and females are also shown, excluding recaptures.

Species	Captures	♂	♀
Bush rat (*Rattus fuscipes*)	62 (123)	23	39
Agile antechinus (*Antechinus agilis*)	20 (38)	5	15
Dusky antechinus (*A. swainsonii*)	8 (10)	4	4
Long nosed bandicoot (*Perameles nasuta*)	2 (4)	1	1
Long nosed potoroo (*Potorous tridactylus*)	1 (1)	0	1
Total	93 (176)	33	60

Significantly more female than male *R. fuscipes* were captured (= 4.14, P <0.05). Similarly, more female *A. agilis* were captured than males (= 5.05, P <0.05). There was no difference in the weights (mean ± se) of male ($\bar{x}$ = 125.9 ± 8.59 g) and female *R. fuscipes* ($\bar{x}$ = 119.8 ± 4.59 g) ($t_{(60)}$ = -0.67, P = 0.49). There was also no difference between the weights of male ($\bar{x}$ = 21.8 ± 0.97 g) and female *A. agilis* ($\bar{x}$ = 24.5 ± 2.52 g) ($t_{(18)}$ = 0.61, P = 0.54), nor between male ($\bar{x}$ = 50.3 ± 20.58 g) and female *A. swainsonii* ($\bar{x}$ = 45.0 ± 2.89 g) ($t_{(6)}$ = 0.25, P = 0.81). Insufficient numbers of the remaining two species were captured to enable meaningful analyses.

The minimum number of small prey animals known to be alive and the composition of prey species varied between sites. The populations of small mammals at site A declined over the study while at sites B and D the populations increased (Figure 16.)

Only *R. fuscipes* provided sufficient captures and recaptures to enable density estimates to be calculated. During the December 2009 trapping session on site B, four individuals were captured, but none of these were recaptured during the trapping session. As a result, the standard error of the density for that trapping session could not be calculated. The population of *R. fuscipes* showed a declining trend at site A, no clear trend at site C, and a gently increasing trend on sites B and D (Figure 17).

All female *A. agilis* captured at site D during September 2010 had hairless pouch young 1 – 1.5 cm in length. The longer pouch young had distinct dark circles where the eyes were developing but had not developed obvious ears. A single female *A. agilis* without pouch young was captured at site C during that same trapping session. This female still had a pouch present with extended nipples, indicating that young were being fed in a nest. Juvenile male *A. swainsonii* had entered the trappable population by the November / December 2009 trapping session.

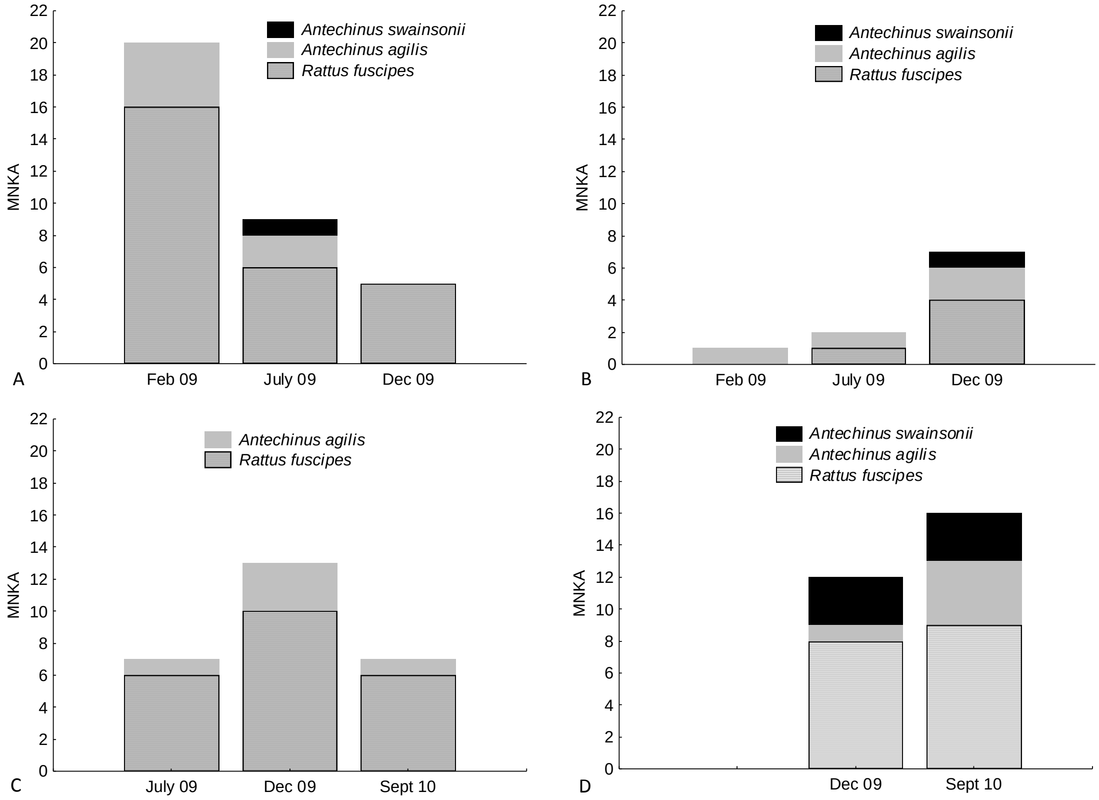


Figure 16. Minimum numbers of animals known to be alive on each of the trapping grids in Far East Gippsland. Grid identifier is shown at the bottom left corner of each graph. Note: Grid D was trapped on two occasions only. Mid-sized mammals are not shown due to low capture rates. Note: trapping session noted as Dec 09 in this figure commenced on 27th November 2009 and continued into December 2009. It is shown as Dec 09 in the above figure only for the sake of brevity.

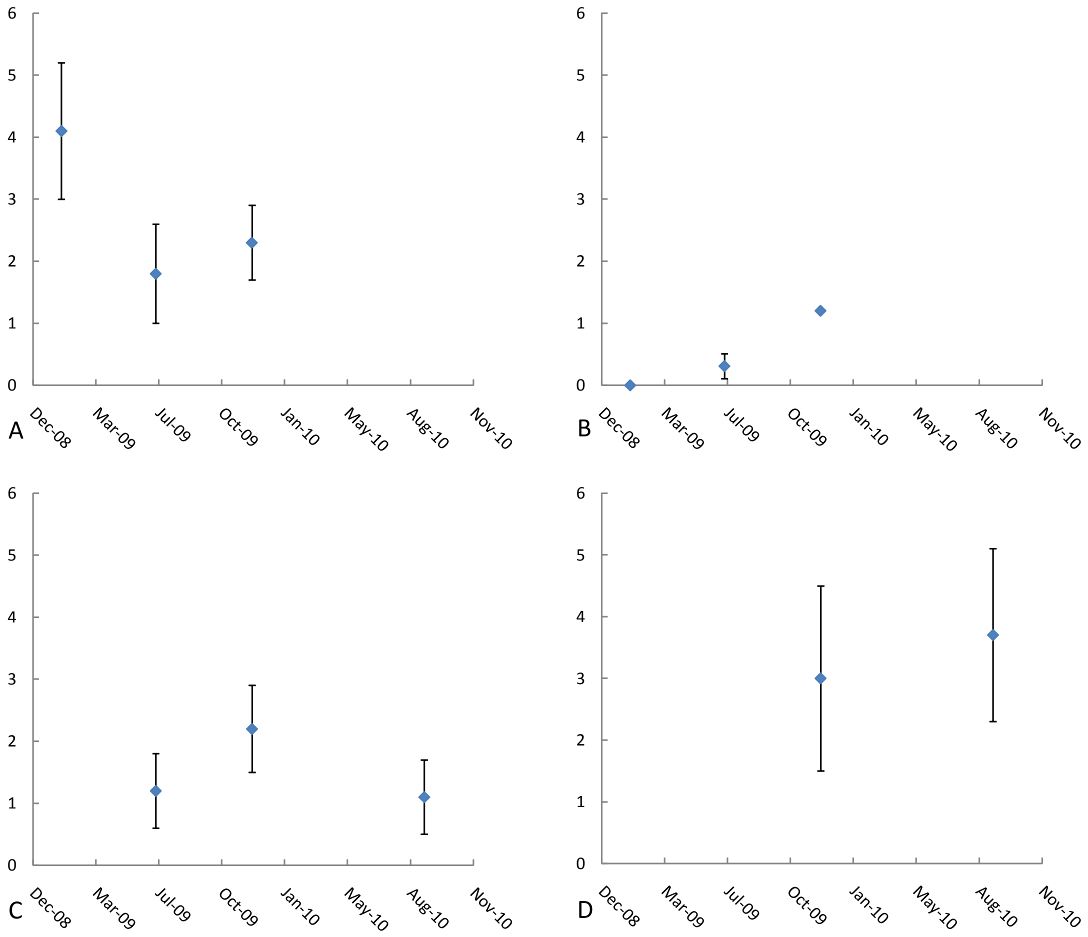


Figure 17. Density of *Rattus fuscipes* at each of four sites in Far East Gippsland. Y axis is density of bush rats ha^{-1}. Site identifier is shown on the bottom left corner of each graph. Density estimates were calculated using the program Density (Efford 2004). Error bars indicate standard error of the density estimate. Note: No recaptures occurred during the December 2009 trapping session on site B. As a result no error bars could be calculated for the population density.

Discussion

The most common species captured during this research, *R. fuscipes* was also the most common prey item in the diet of feral cats in Far East Gippsland (Appendix 1). However, the agile antechinus, *A. agilis*, was the second most commonly captured mammal yet was not found in the diet of the feral cats. Fewer *A. swainsonii* were captured than *A. agilis* yet that species appeared as a dietary component in nearly 20% of the cat scats analysed. The scansorial nature of *A. agilis* may assist it to avoid predation more readily than the wholly terrestrial *A. swainsonii* but, if such behaviour is advantageous, it does not account for why the predominantly arboreal ringtail possum, *Pseudocheirus peregrinus* (not captured in this study), was found in about 10% of cat scats. As *A. agilis* (reported as *A. stuartii*) was found by Triggs *et* *al.*.(1984) in 15% of cat scats, it is possible that the low number of scats analysed in Appendix 1 may have introduced bias into the dietary analyses. However, it is also possible that the lower body mass of the *A. agilis* makes them energetically unprofitable to pursue when compared with the larger *A. swainsonii*, especially when the latter species and the even larger bush rats are available. The paucity of captures of mid-sized mammals unfortunately precluded any assessment being made of their distribution or abundance in the forest. Neither the long nosed bandicoot, *Perameles nasuta*, nor the long nosed potoroo, *Potorous tridactylus*, were detected in the diet of cats in the present study (Appendix 1). It is possible that these species are too scarce, or too large (adults in both species achieve body weights >1 kg) to be hunted regularly, but further work is needed to evaluate these possibilities.

The *R. fuscipes* population exhibited variable trends over the period of the study. A declining trend was found at site A, no clear trend at site C, while increasing trends were found at sites B and D. As there was considerable overlap in the standard errors of the density estimates within each site, it is possible that the temporal patterns in density were more apparent than real. If the patterns did reflect actual population trends, it is unlikely that they were influenced by fox baiting by Southern Ark as baiting had occurred in the vicinity of sites C and D prior to my study but did not occur at sites A and B (Diment 2010; Southern Ark, unpublished data). Instead, the forests of Far East Gippsland are highly, but patchily, productive (Cameron *et al.* 2005), and this may provide a more likely explanation for the trends observed. Thus, the population fluctuations of *R. fuscipes* may reflect site-specific responses as animals tracked locally different changes in phenology or other resources over the course of the study (Lindenmayer *et al.* 2005; Taylor and Calaby 1988; Wood 1971).

The higher numbers of female than male *R. fuscipes* captured during this study differ markedly from the results of most other studies (e.g. Robinson 1988; Taylor and Calaby 1988; Wood 1971). Generally the ratio of males to females is closer to 50:50 or male biased, rather than showing the female dominance as found in this study. However, Wilson *et al.* (1986) found that the ratio of male to female *R. fuscipes* varied throughout the year with periods when only females were captured and others when captures of males outweighed those of females 2:1; overall, there were more females captured than males. It is probable that the female bias found in this study is a result of the natural fluctuation of the *R. fuscipes* population, with my sampling taking place when females were either more abundant or more readily trappable.

Many of the early records of *A. agilis* in Victoria were recorded as *A. stuartii*. This species was originally classified taxonomically as *A. stuartii* until separated as a distinct and separate species (Dickman *et al.* 1988; Dickman *et al.* 1998). While both species are nearly identical morphologically, they have predominantly separate distributions with limited overlap at Kioloa on the south coast of New South Wales and in parts of the southern tablelands of that state. *Antechinus agilis* occurs generally to the south of Kioloa and *A. stuartii* to the north (Menkhorst and Knight 2001), with no confirmed records of the latter species from Victoria.

The capture of more female than male *A. agilis* in my work most likely arose from the timing of the sampling. As noted, members of the genus *Antechinus* are semelparous; all males die following a brief but intense period of mating in mid to late winter (Dickman 1980; Dickman 1985; Taylor and Horner 1970). Males generally do not enter the trappable population until mid to late summer the following year (Friend 1985; Smith 1984). Sampling in this study took place mainly during the periods following male die-off and prior to juvenile males entering the trappable population.

The lack of sexual dimorphism that I found in the two *Antechinus* species captured here may also result from the timing of the trapping. Strong sexual dimorphism is present in all *Antechinus* species, with adult – but not juvenile or subadult – males being 50-100% heavier and larger than females. While all species are sexually promiscuous, the weight of the male is correlated positively with the number of females fertilised and the number of offspring produced (Kraaijeveld-Smit *et al.* 2003). It is most likely that trapping of adults prior to the breeding season would have revealed much stronger sexual dimorphism in these species than my results would otherwise suggest.

The breeding season for most *Antechinus* species varies primarily depending on the rate of change of photoperiod (change in daylight length) in spring (McAllan and Dickman 1986; McAllan and Geiser 2006), although other factors such as population density may also play a role (Dickman 1985). For *A. agilis* and other species in the temperate and subtropical zones, there is a distinct correlation between latitude and timing of breeding (McAllan and Dickman 1986; McAllan *et al.* 2006; Watt 1997).

The pouch young in a female *A. agilis* captured in September 2010 (26th – 28th) were hairless and 1 – 1.5 cm long. Marlow (1961) examined the growth rates of the larger *A. stuartii* (reported as *A. flavipes*) and indicated that pouch young of that size were between 12 and 20 days old. *Antechinus agilis* is a smaller species than *A. stuartii* and as a result, the pouch young found in this study are potentially older than would be predicted from Marlow (1961). However, as no data are available in the literature regarding the growth rates of *A. agilis*, I will use that proposed by Marlow (1961) as a minimum age. As the gestation period for *A. agilis* is 27 days (Selwood 1980), conception occurred between the 9th and 17th August. This places the breeding period in early to mid August for *A. agilis* in Far East Gippsland at the latest. This is slightly earlier than the time predicted using the model of McAllan and Dickman (1986) who found that the critical rate of change in photoperiod that triggers breeding is +127 to +137 seconds per day. This rate of change occurs from the 18th August to 1st September at the latitude of my study (37°.32′ S) (Figure 18). When in sympatry with *A. swainsonii*, as in this study, *A. agilis* tends to breed later than when in allopatry (Dickman 1982).

The breeding season of *A. agilis* in this study coincides with that of *A. agilis* in the Otway mountains ranges in western Victoria which lie between 1 and 1.5 degrees latitude further south (Scott 1986). As the Otway ranges are further south it would be expected that the breeding season at my sites would occur later than that of the animals in the Otways rather than in synchrony (McAllan and Dickman 1986).

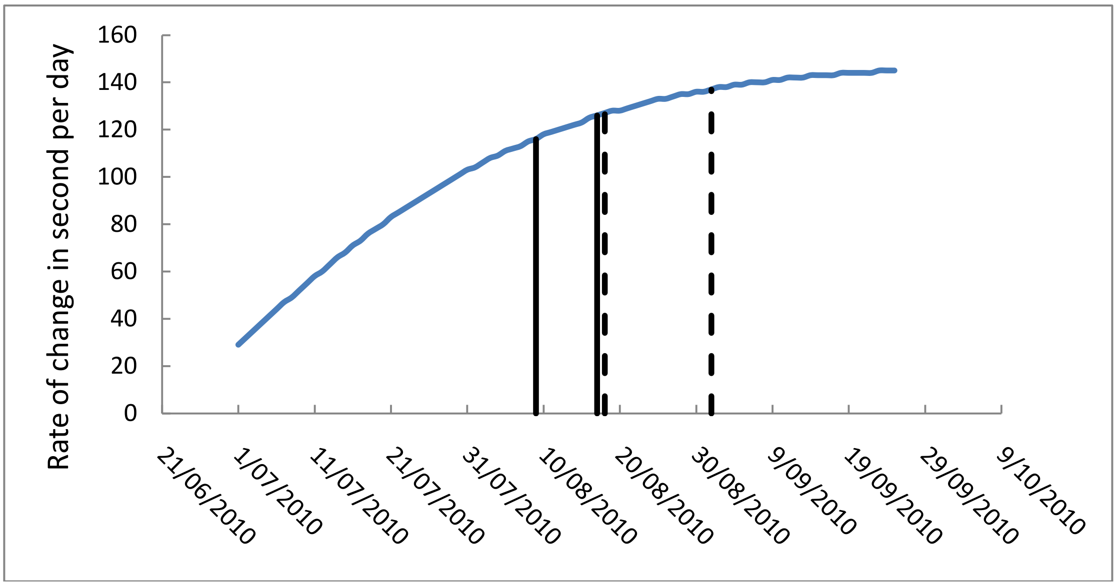


Figure 18. Rate of change in photoperiod for latitude 37°32′ South. Y axis shows the rate of change in photoperiod in seconds per day. Dashed vertical lines show the period when the rate of change increases from 127 to 137 seconds per day which is the predicted breeding period for *A. agilis*. Solid lines show the probable breeding period found in this study. It is noted that the actual breeding period would last longer than indicated here. Adapted from McAllan and Dickman (1986) and day length data from the Bureau of Meteorology for Melbourne.

The single *A. agilis* captured without pouch young yet still having extended nipples tends to indicate that either the young were being suckled in a nest, or that she had recently ejected the pouch young prior to being captured. No ejected pouch young were found in or near the trap in which she was captured. Juvenile antechinus attach very firmly to the mother's teats for 36 – 40 days from birth (Marlow 1961; Smith 1984). Following that time, the young are left in a nest while the mother hunts. If the single *A. agilis* captured without pouch young had deposited her young in a nest by the time of capture, it would tend to indicate the breeding season for the population begins in late July.

McAllan and Dickman (1986) suggested that there might be isolated remnant populations of *A. stuartii* in certain areas in Victoria. This species breeds when the rate of change of photoperiod is between 97 and 107 seconds per day. Isolated populations of *Antechinus* have been found at Mallacoota, Frankston and in the Grampian Ranges that breed when that rate of change of photoperiod is achieved. This rate of change occurs at my study sites in late July. If the single *Antechinus* captured had indeed deposited her young into a nest at the time of capture, and if the advanced state of growth of the young of other females is considered, it is possible that this may be another isolated population of *A. stuartii* rather than *A. agilis*.

Alternatively, no *A. swainsonii* were captured at site C during this study, nor were any recorded during a 7-year trapping program by Parks Victoria (R. Korn, Parks Victoria, pers. comm.) that immediately preceded my study. This indicates that *A. agilis* in this area are allopatric, in which case, breeding by *A. agilis* would tend to take place earlier than when in sympatry with *A. swainsonii* (Dickman 1982) such as at the other three sites trapped in the study. Clearly, the potential for an earlier breeding season in *A. agilis* in Far East Gippsland than in other populations at similar latitudes warrants further investigation. I note that the sample size in the present study is low and that definite conclusions should be drawn with caution. Nonetheless, as animals appeared to be breeding earlier than expected, this may have considerable bearing on when poison baiting for cats could be safely carried out, and should be investigated in more detail to determine exactly when breeding occurs.

Juvenile male *A. swainsonii* had entered the trappable population by the end of November 2009. Juvenile *Antechinus* enter the trappable population about five months after the breeding period. This indicates that breeding of *A. swainsonii* in Far East Gippsland occurs in late June at the latest. This is up to one month earlier than occurs at other locations at similar latitudes in Victoria, for example, in the Otway ranges and at Sherbrooke (Cockburn *et al.* 1985). Juveniles did not enter the trappable populations until mid to late December at these locations. Dickman (1982 p. 146) suggested that altitude rather than latitude is one of the primary factors in the timing of breeding in *A. swainsonii*, with animals at lower altitudes breeding earlier. Timing of the breeding season in Far East Gippsland may be earlier due to the low elevation at the study sites (160 m AMSL). A further contributing factor may well be that populations there are sympatric with *A. agilis*, which could also result in an earlier breeding period than if the population was allopatric (Dickman 1982).

If these differences in the breeding seasons of *Antechinus* spp. in Far East Gippsland have been interpreted correctly, they may require alterations to current management practices in that region. Trapping was scheduled in this study to avoid times when young were in the pouch or had just been placed in the nest. This was done to avoid the potential for pouch young to be ejected due to stress on the mother while in the trap. Similarly, it was anticipated that the trapping schedule would avoid times when young had left the pouch and had been placed in the nest but were still dependant on suckling the mother for food. Restraining the mother in a trap for a period of time may reduce the survival of the nest young. Regular trapping as part of the Southern Ark project should be avoided at times when young *Antechinus* have just been placed in the nest, as restraining the mother in a trap for extended period may impact negatively on the nest-bound juveniles. It is recommended that a survey of the breeding period of *Antechinus* species in Far East Gippsland be undertaken to establish the temporal variability of the breeding season in that region compared with the rest of the state. Additionally, it is recommended that additional trapping of the *Antechinus* populations takes place at the Coopracambra National Park site to determine which species is present.

Feral cats generally eat carrion, and therefore baits, only at times when they are food stressed. In the south eastern states of Australia this is generally in late winter and early spring when food resources are at a minimum, for example following the male die-off in *Antechinus* spp. If an earlier breeding season for *Antechinus* causes a temporal shift in the timing of the food shortage, it therefore changes the best time to lay baits for cats. Based on this research, the optimal time for baiting feral cats is between the die off of male *A. agilis* in late August and the time that juvenile male *A. swainsonii* enter the trappable population at the end of November. The bush rat population would also be at a low ebb at this time. Poison baiting for feral cats using baits where the toxicant in encapsulated within a hard shelled delivery vehicle (HSDV) that maintained structural integrity would result in little or no adverse effects on the remaining females (Chapters 7 and 8).

Poison baiting at this time using surface-laid baits into which the toxicant has been directly injected is not recommended. The female *Antechinus* are lactating at this time which increases their metabolic requirements. As many insects are in diapauses at this time, it would be expected that the females, like the feral cats, would be food stressed and more likely to consume poisoned baits. If baiting at this time with directly injected poison baits was a necessity, it is recommended that the bait be suspended above ground and beyond the reach of female antechinus using a technique similar to the gantry employed by Algar and Brazell (2008) on Christmas Island.

Chapter 7

Potential for ingestion by non-target Australian animals of Curiosity® baits with toxicant enclosed in a hard shell delivery vehicle – a desktop analysis

Chapter overview

Effective population control of feral cats remains an almost intractable problem, with very few techniques available for use by managers. Poison baiting is used over large areas in Western Australia to manage feral cat populations, with sodium monofluoroacetate (compound 1080) used as the primary toxicant. However, while many native mammal and bird species have a natural tolerance to 1080 in western and northern regions of Australia, their eastern conspecifics do not. Because of the potential risks of poisoning to non-target species, this precludes the laying of 1080 baits on the ground surface in the eastern states of Australia. Trials are currently underway to determine if encapsulating the toxicant within a hard shelled delivery vehicle is effective in minimising access to the toxicant by non-target species.

Here I present the results of a desktop analysis used to investigate the likelihood of Australian vertebrate animals consuming Curiosity® baits, or similar media for baiting feral cats, and the potential of these animals to access toxicants that may be incorporated into the baits using a hard shell delivery vehicle (HSDV) approach. The HSDV comprises toxicant compound encapsulated in an acid soluble polymer approximately 6 mm in diameter (Hetherington *et al.* 2007; Marks *et al.* 2006) and which is inserted into the Curiosity® bait media. I compare the numbers of animals that have the potential to consume just the Curiosity® bait with those that have the potential to consume the HSDV to determine if incorporating HSDVs can reduce the likelihood of poisoning of non-target species.

The four main groups of vertebrate land animals: mammals, reptiles, birds and amphibians, were analysed using an *a priori* determined list of criteria. The analysis included 3796 species and subspecies native to Australia as well as accidental and occasional visitors.

Most species would not be exposed to the toxicant if surface baits are used to control feral cats. Many of the species examined are either too small to consume the bait or have diets that would usually preclude them from eating food such as the bait. Of those species that would consume the Curiosity® bait, significantly fewer would be susceptible to non-target poisoning if HSDVs were used as opposed to direct injection of toxicant into the bait media. Carnivorous mammals were the most likely to consume both the baits and the HSDV.

Introduction

The control of invasive predator species for the conservation of biodiversity and protection of primary production assets is conducted principally through the use of poisons (Olsen 1998). However, a primary consideration when using a toxicant for the control of any species is the likelihood that non-target species will take the bait (Calver *et al.* 1989a; Glen *et al.* 2007b). Consumption of poisoned baits by non-target species can have lethal consequences (Dexter and Meek 1998; Hetherington *et al.* 2007; Martin *et al.* 2002), and excessive bait take by non-target species can limit the effectiveness of the control program (Algar and Brazell 2008). Baits are also used for the delivery of oral vaccines for disease control (Olson *et al.* 2000; Steelman *et al.* 2000) and potentially for the dissemination of immunocontraceptives (Miller *et al.* 1999). Ensuring effective bait delivery is clearly imperative in any control campaign.

European foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) have contributed to the extinction of many native mammalian species and have been implicated in the decline of many others in Australia (Burbidge *et al.* 2009; Dickman 1996; Risbey *et al.* 1999; Risbey *et al.* 2000). Because of its acute toxicity to most species of mammals, 1080 (sodium monofluoroacetate) is the only registered toxicant available for the control of invasive predators in Australia (McIlroy 1981a; McIlroy 1981b; McIlroy 1982a). Many native mammal species in western and north-western Australia are relatively tolerant to this toxicant as they coevolved with endemic plant species that contain naturally occurring fluoroacetate compounds (Martin and Twigg 2002; Twigg and King 1989; Twigg and King 1991). Mammals in eastern Australia are less tolerant and are more susceptible to 1080 toxicosis than their western counterparts; as result, 1080 baits laid in eastern Australia need to be buried to minimise the potential for non-target species to access them (Fairbridge *et al.* 2003; Twigg and King 1991). Bait burial provides little deterrence for canid species as their acute olfactory senses allow them to detect buried baits which are then excavated and consumed (Bradshaw 2006).

Feral cats do not possess the same acute olfactory senses as canids and rarely locate and excavate buried baits (Algar and Burrows 2004). They demonstrate a preference for live prey (Leyhausen 1979) over carrion or dried meat baits, and therefore baits intended for feral cats must be surface-laid and consist preferably of a wet meat or similar material (Algar *et al.* 2007; Algar and Burrows 2004). However, surface-laying of baits increases the potential for non-target species to access the bait and consume a lethal amount of toxicant (Glen *et al.* 2007b).

Currently trials are being undertaken to determine the suitability of para-aminopropiophenone (PAPP) as a substitute toxicant for some control applications (Fisher *et al.* 2005; Johnston *et al.* 2010a; Johnston *et al.* 2011; Murphy *et al.* 2007). In many instances, LD_{50} rates - the amount of toxicant needed to kill 50% of the sample group - for PAPP are far higher than those for 1080. For example, the LD_{50} for PAPP for *F. catus*, is 5.56 mg/kg (Savarie *et al.* 1983) while for 1080 it is between 0.28 mg/kg (Eason and Frampton 1991) and 0.4 mg/kg (McIlroy 1982b). Having a lower toxicity, non-target species must consume more of the PAPP to obtain a lethal dose than would be required if using 1080.

Encapsulation of the toxicant (both 1080 and PAPP) within bait media is also being trialled as a means of minimising the exposure of non-target species to the toxicant (Hetherington *et al.* 2007; Johnston *et al.* 2009; Marks *et al.* 2006). The toxicant is sealed within a small, hard, acid-soluble polymer capsule known as a hard shelled delivery vehicle (HSDV). The HSDV is then inserted into the bait prior to it being laid. Many Australian native animals gnaw at baits rather than consuming them with a single, or several, large bites (Hetherington *et al.* 2007) (Figure 19 and Figure 20). When a native animal encounters the hard capsule while gnawing the bait, it may be rejected (i.e. spat out) rather than consumed (Hetherington *et al.* 2007; Marks *et al.* 2006).

Approving authorities usually require extensive field and pen trials to determine which non-target species are likely to take baits (Brunner 1983). Here I use a desktop analysis incorporating appropriate parameters based on the biology of all native Australian species to determine which non-target animals are likely to take baits. This can reduce the number of species that need to be tested. I use the best current knowledge for each species to determine the parameters, including what may appear to be aberrations of behaviour such as foxes being poisoned through consuming fermented wheat baits for pigs (Twigg *et al.* 2007). Parameters must be updated as further research findings become available.

Methods

Bait and capsule

The Curiosity® bait (Johnston *et al.* 2009; Johnston *et al.* 2011) was chosen as the bait medium for this analysis (Figure 21). The Curiosity® bait is essentially the Eradicat® cat bait manufactured by the Department of Environment and Conservation in Western Australia (Patent number AU 781829) but containing a HSDV rather than directly injected 1080 (Johnston *et al.* 2011). It is a chipolata sausage-style wet meat based bait medium ~15 g in weight and 10 cm in length consisting of kangaroo meat, chicken fat, digests and flavour enhancers (Algar and Burrows 2004; Hetherington *et al.* 2007). The Curiosity® bait is also buffered slightly differently to the Eradicat® to ensure HSDV stability within the bait. The HSDV is a proprietary product manufactured by Scientec Research Pty Ltd (Melbourne, Victoria) designed specifically to encapsulate the PAPP toxicant (Provisional Patent No. 200890357). It is ~6 mm in diameter and formulated to dissolve swiftly in stomach acid, thus releasing the toxicant into the stomach rapidly following consumption (Johnston *et al.* 2009; Johnston *et al.* 2008).

Figure 19. Image of brush-tailed possum (*Trichosurus vulpecula*) gnawing an Eradicat® bait. Photograph: T. Buckmaster

Figure 20. Image of European fox (*Vulpes vulpes*) consuming an Eradicat® bait in several large bites. Photograph: T. Buckmaster

Figure 21. Eradicat® feral cat bait (left) and HSDV (right) with Rhodamine B bait marker enclosed within. Photograph: T. Buckmaster

Species list

Comprehensive species lists of all Australian terrestrial vertebrate animals were obtained from the Department of the Environment, Water, Heritage and the Arts, Australian Biological Resources Study (ABRS) for the four taxonomic classes being analysed: Amphibia (DEWHA 2009a), Aves (DEWHA 2009b), Mammalia (DEWHA 2009c), and Reptilia (DEWHA 2009d). These lists of Australian fauna are publicly accessible from the ABRS web site at (http://www.environment.gov.au/biodiversity/abrs/index.html) and are assumed to provide the current scientific name for each species and subspecies.

These species data files were downloaded directly from the ABRS website as CSV files and then converted into MS Excel spreadsheets. Superfluous data (for example: species synonyms, details of amendments to species common and scientific names and historical nomenclature) were removed. The remaining data included for each species: common name, current scientific name and recognised subspecies. All subsequent analyses were undertaken using these modified lists.

Information on animal size, habits, diet, conservation status and other aspects relevant to this desktop review was sought from a variety of field guides, checklists and other published sources. Due to the continual and ongoing revision and variation in the taxonomy of Australian vertebrates, however, most references used for the analysis inevitably contained outdated nomenclature for some species and subspecies. Where this was encountered, the original ABRS CSV file was checked to ensure that the species referred to in the literature was the same as the species on my lists. The final analysis was undertaken on species lists as they stood with the ABRS as of the 3rd March 2009: subsequent taxonomic revisions of Australian vertebrates have not been included in this analysis. Recently Christidis and Boles (2008) undertook a review of the systematics and taxonomy of Australian birds. As the ABRS has adopted the nomenclature of that review, and as my analysis follows the ABRS, it is acknowledged that it may differ from that used in some currently available species guides.

In several instances the ABRS database listed animals that were not present in any guide used in this analysis. A search of the recent literature was used to obtain information on the morphometrics and other biological characteristics of these species. Generally discrepancies arose when a previously described species had been reclassified into multiple new species for which no description had been published outside the scientific literature. In these instances, details of the species were obtained from the relevant literature.

Common names for species have been taken from the ABRS data files. There may be instances where a locally used common name does not conform with that listed by the ABRS. No attempt was made to determine all possible common names for species used in this analysis.

Species attributes

Size of animals

The maximum weight for each mammal is listed in grams; in cases where maxima were listed but qualified by the reference with a usual weight range, I used the maxima for the usual weight range. Mammalian weights were sourced from Menkhorst and Knight (2001). The length of birds, inclusive of tail is listed in centimetres and was taken from Pizzey and Knight (1997). Lengths for amphibians and reptiles are as given in Cogger (2000) or Wilson and Swan (2008). Combined body plus tail length of snakes and turtles was listed while snout – vent length was listed for the remaining reptiles.

Conservation status

Conservation status for each species was obtained at the Commonwealth level from Clayton *et al.* (2006). Whilst listings are available in this reference for each state and territory, only the Commonwealth conservation status is used in this analysis.

Potential for bait or HSDV consumption

No field or pen testing of any animal was conducted for this analysis. Bait-take potential was assessed instead for each species using decision tree analysis (Dickman *et al.* 2010) with assessment criteria listed in Appendix A of this chapter. These criteria were developed to assess each species' potential to consume the bait, and included diet, feeding behaviour and size of the animal. For HSDV consumption the criteria included potential for bait consumption, body size and diet and the results, if any, of pen and field studies. Data from published and unpublished reports from field and pen trials could be, in some instances, extrapolated to morphologically similar species. Relevant information for the selected criteria was taken from the morphometric, dietary and habitat preferences as listed in the species guides used (mammals - Menkhorst and Knight (2001), amphibians and reptiles – Wilson and Swan (2008) and Cogger (2000), and birds - Pizzey and Knight (1997)). For most birds and reptiles, dietary and habitat information was listed only at the family level so the assessment was undertaken at that level. Where field or pen trails had been undertaken reference is made to those trials and an assessment based on those trials is made (for example Marks *et al.* (2006) and Hetherington *et al.* (2007)).

The probability of unintentional consumption of the bait media was not considered in this analysis. For example, a large obligate herbivore would not actively seek out and consume meat-based baits, but may unintentionally consume one while foraging. For this to be quantified with any degree of certainty, field or pen testing needs to be undertaken and was beyond the scope of this analysis. I have not consistently assessed the possibility of toxicants leaking into the bait media after degradation of the capsule, but note any studies that have shown this to occur.

Where possible, I make recommendations to minimise consumption of the bait media by listed species based on the available biological data. A recommendation is made for the future testing of any species if it was apparent that there was potential for consumption of the bait media or HSDVs by those particular species. No attempt is made to prioritise one species for testing ahead of another.

Chi squared analyses, with Yates' correction where appropriate, were used to compare the numbers of non-target species likely to be exposed using toxicant injected into bait media with those at potential risk from toxicants enclosed within HSDVs.

Results

In total, 3 769 vertebrate species and subspecies were examined in this review, and 490 species were determined to either consume or have the potential to consume the Curiosity® bait media if deployed on the ground surface rather than as buried baits. The analysis revealed that, of these, only 47 species would consume the HSDVs of encapsulated toxicant within the bait media while a further 343 species potentially may consume the HSDV (Table 15). A complete list of all species used in this analysis and the complete results listed by taxonomic class is attached in electronic form as Appendix 2 to this thesis.

Table 15. Numbers of species in each taxonomic class used in the analysis and the numbers within each taxonomic class assessed to have the potential to consume bait media and have access to the encapsulated toxicant.

Group	No. analysed	Bait		HSDV	
		Will consume	Possibly consume	Will consume	Possibly consume
Mammalia	582	157	20	21	69
Aves	1872	24	239	12	239
Reptilia	1086	40	10	14	35
Amphibia	229	0	0	0	0
Total	3769	221	269	47	343

Only a single species on the Environment Protection and Biodiversity Conservation (EPBC) Act 1999 list, the Tasmanian devil (*Sarcophilus harrisii*) was assessed as definitely consuming the Curiosity® bait and enclosed HSDV. A further 21 species on the EPBC list were found to have potential to consume the HSDV (Table 16).

Significantly fewer non-target species are susceptible to being exposed to the toxicant when using a HDSV within the bait media (= 140.7, P <0.001) than if the toxicant was injected directly into the bait (Figure 22). At a taxonomic class level, other than for Amphibia, the use of HSDVs markedly decreased the number of species susceptible to non-target poisoning – Mammalia (= 913.7, P <0.001), Aves (= 12.1, P <0.001) and Reptilia (= 48.9, P <0.001). No amphibians were assessed as being likely to consume the Curiosity® bait media so non-target poisoning of amphibians was therefore considered unlikely.

Table 16. Species listed under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* that have the potential to consume HSDVs during a ground surface baiting campaign. E = Endangered V = Vulnerable.

Common Name	Scientific Name	Status
Southern cassowary	*Casuarius casuarius*	E
Malleefowl	*Leipoa ocellata*	V
Antarctic tern subsp. bethunei	*Sterna vittata bethunei*	E
Antarctic tern subsp. vittata	*Sterna vittata vittata*	V
Pied currawong subsp. crissalis	*Strepera graculina crissalis*	V
Brush-tailed bettong subsp. oglibyi	*Bettongia penicillata ogilbyi*	E
Crest-tailed mulgara	*Dasycercus cristicauda*	V
Ampurta	*Dasycercus hillieri*	E
Kowari	*Dasyuroides byrnei*	V
Western quoll (Chuditch)	*Dasyurus geoffroii*	V
Northern quoll	*Dasyurus hallucatus*	E
Spotted-tailed quoll	*Dasyurus maculatus gracilis*	E
Spotted-tailed quoll	*Dasyurus maculatus maculatus*	E / V
Golden bandicoot subsp. auratus	*Isoodon auratus auratus*	V
Golden bandicoot subsp. barrowensis	*Isoodon auratus barrowensis*	V
Dibbler	*Parantechinus apicalis*	E
Eastern barred bandicoot unnamed subsp.	*Perameles gunnii* subsp. (Victoria)	E
Eastern barred bandicoot subsp. gunnii	*Perameles gunnii gunnii*	V
Red-tailed phascogale	*Phascogale calura*	E
Northern brush-tailed phascogale	*Phascogale tapoatafa pirata*	V
Tasmanian devil	*Sarcophilus harrisii*	E
Julia Creek dunnart	*Sminthopsis douglasi*	E

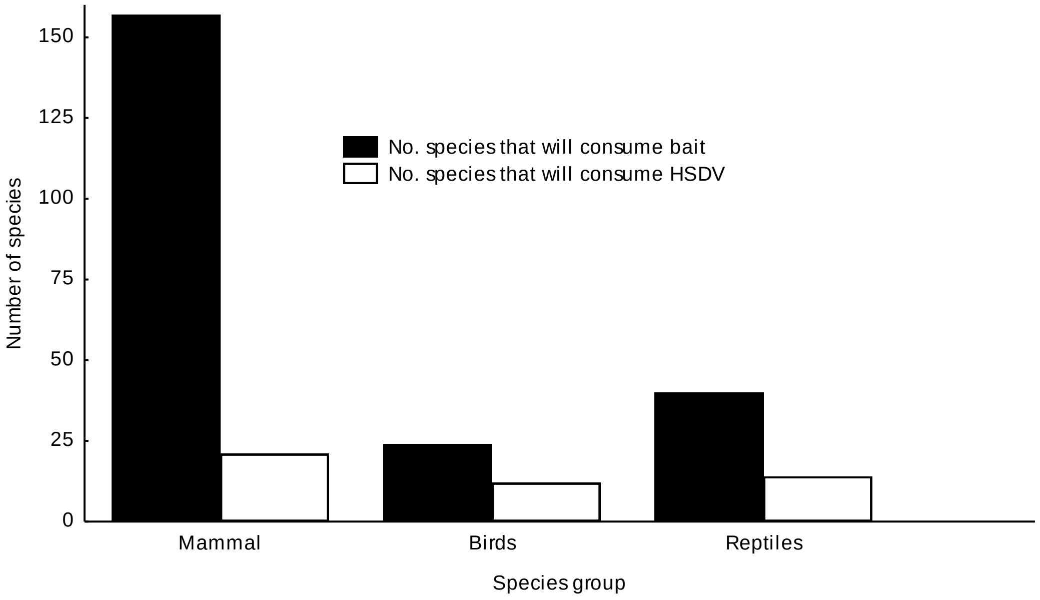


Figure 22. Number of species / subspecies assessed to be susceptible to poisoning if toxicant is injected directly into the Curiosity® feral cat bait or if toxicant is enclosed in a HSDV within the bait.

Discussion

This analysis revealed that most Australian vertebrates would not be susceptible to non-target poisoning if surface-laid baits are used during programs to control feral cats. Significantly fewer non-target species would access the toxin when enclosed within HSDVs than would occur if the toxicant was injected directly into the bait media, as is the current practice using 1080. Similarly, in pen trials, Hetherington *et al.* (2007) showed that using a HSDV would reduce the potential impacts of a poisoning campaign on Western quolls (*Dasyurus geoffroii*), Brush-tailed bettongs (*Bettongia pencillata*) and Southern brown bandicoots (*Isoodon obesulus*) while Marks *et al.* (2006) found that a HSDV would reduce the ability of Northern quolls (*Dasyurus hallucatus*) to access toxicants within feral cat baits.

Mammals, particularly carnivorous marsupials, were assessed as the most susceptible group for consuming the bait and the enclosed HSDV. However the number of species likely to consume the HSDV was significantly lower than those likely to consume the bait. Carnivorous animals, particularly marsupials, are the most likely to be susceptible to non-target poising when using the Curiosity® bait as the bait is designed to target a carnivorous mammalian pest – the feral cat (Algar and Burrows 2004; Hetherington *et al.* 2007).

The lack of amphibians susceptible to non-target poisoning reflects the small size of the Australian amphibians when compared with the size of the bait. Similarly, most birds were excluded as non-target species due to their size in comparison with the bait.

My analysis did not consider the possibility of the HSDV decaying prior to being consumed by an animal. Such decay may take two forms. Firstly, the capsule may decay at the folds on the end and, while still retaining structural integrity, toxicant may leach into the bait media. Secondly, when the capsule softens from contact with the bait medium, it is likely to be cracked or punctured with far less effort than if it had retained its structural integrity. Either form of decay would result in the encapsulated toxicant becoming available to non-target species. As I show in Chapter 8, HSDV decay within the bait media is possible. By manipulating the manufacture process of the bait, Scientec Research Pty Ltd is in the process of minimising the potential for HSDV decay within the bait media (M O'Donoghue, Scientec Research Pty Ltd, pers comm).

By examining the behaviour and feeding patterns of possible non-target species it should be possible to minimise the potential for bait-take by these species still further. Varanids, for example, are very active during the warmer months and have the potential to consume Curiosity® bait media and the HSDV. By shifting a baiting campaign to late winter, most varanids will be in hibernation and are therefore unlikely to consume the bait. Algar and Burrows (2004) indicated that the most likely time feral cats would take baits is when they are food stressed. During the winter months in the eastern states of Australia, many of the food sources of feral cats decline (e.g. *Antechinus* males die off following the breeding season (Banks *et al.* 2005; Dickman 1980), reptiles are less active (Wilson and Swan 2008) and most insects are in diapause or overwintering as eggs or pupae). Targeting feral cats at this time would both increase the likelihood of cats taking baits and minimise the uptake by varanids.

Many of the non-target mammal species that are likely to consume baits are small (<200 g). Suspending the baits above the ground would remove them from the reach of these animals and greatly minimise the possibility of non-target bait take. Algar and Brazell (2008) devised a gantry device to suspend the Curiosity® baits out of the reach of both black rats (*Rattus rattus*) and land crabs (*Cardisoma carnifex*). The device minimised non-target problems on Christmas Island by suspending the bait 40 cm above the ground, beyond the reach of the non-target species yet still within the reach of feral cats (Algar and Brazell 2008; Johnston *et al.* 2009).

Many of the difficulties with using surface-laid baits for pest or feral animals can be reduced by encapsulating the toxicant within a HSDV. Encapsulation within the HSDV would allow other toxicants such as 1080 that are not able to be used in surface-laid baits to be now considered for use. However, while every effort needs to be taken to minimise the poisoning of non-target species during feral cat control programs and using a HSDV for toxicants can minimise the risk of non-target deaths, it cannot completely negate it. All intervention in wildlife management carries some risk to both target and non-target species so a considered stepwise approach to managing and addressing risk needs to be undertaken.

The analysis presented here shows the potential to minimise the exposure of non-target species to a toxicant by using a HSDV. However the actual degree to which many of the species listed as potentially taking the bait media / HSDV needs to be tested through pen and field trials. It is also acknowledged that the poisoning of non-target species may still occur in unexpected circumstances not considered in this analysis.

I have shown that a decision tree process can provide a transparent and repeatable assessment method for identifying animals that are likely to take baits and / or HSDV. This approach also allows the incorporation of new data within the decision making process when it become available. A similar process was used by Dickman *et al.* (2010) to prioritise areas in Australia for feral cat control. The decision making process utilised in this chapter can be adapted for most bait types, making it a useful tool for preliminary risk assessment of taxa that may otherwise be non-target victims in poison baiting campaigns.

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Appendix A to chapter 7

Criteria for assessment of the likelihood that Australian terrestrial vertebrates will consume sausage style meat baits (Curiosity®) designed for feral cats (A1) and toxins encapsulated in HSDVs within the bait (A2)

Table A1: Potential for consumption of bait substrate. Assessment is modified by each subsequent level. For example a carnivore that feeds predominantly at sea will be assessed as having no potential to consume the bait. When a "No" assessment is made, analysis for that animal ceases.

Criteria	Assessment of potential
A. Diet	
Carnivore / scavenger (including scavenging omnivores)	Yes – unless otherwise shown by lab / field testing
Terrestrial insectivore	Yes – unless otherwise shown by lab / field testing
Omnivore	Probable – unless otherwise shown by lab / field testing
Terrestrial foraging hypogeal fungivore	Probable– unless otherwise shown by lab / field testing
Herbivores / frugivores / insectivores / exudate feeders: including specialist or obligate feeders	No
Volant insectivores	No
Granivorous birds	No
B. Feeding behaviour	
Feeds predominantly at sea	No
Specialist ant feeders	No
Where foraging activities are unlikely to result in baits being identified as food sources	No
Unlikely to recognise bait as food source e.g. cursorial carnivores with minimal scavenging	No
C. Size	
Birds / amphibians / reptiles too small to consume bait*	No

*For this category, birds < 26cm, amphibians < 15cm and reptiles < 30cm SVL or 50 cm in total length were assumed to be too small to eat the baits.

Table A2: Potential for consumption of HSDV. Assessment is modified by the subsequent level. For example a large carnivore that has demonstrated complete HSDV rejection in field or pen studies will be assessed as having no potential to consume the HSDV. When a "No" assessment is made, analysis for that animal ceases.

Criteria	Assessment of potential
A. Bait consumption	
Where potential for bait consumption = No	No
B. Size and diet	
Large eutherian carnivore*	Yes
Carnivore / insectivore larger than smallest known to consume HSDV (Determination based on body size and feeding habits)	Possible /Yes
Carnivore / Insectivore smaller than smallest known to consume HSDV (Determination based on body size and feeding habits)	Possible / No
Omnivore larger than smallest mammal known to consume HSDV with similar feeding habits to those known to consume HSDVs	Individual determination
Omnivore smaller than smallest known to consume HSDV	No
C. Field and pen studies	
Complete HSDV rejection confirmed by lab / field experiment	No
Predominant HSDV rejection demonstrated by lab / field experiment	Unlikely with data shown
Predominant HSDV consumption demonstrated by lab / field experiments	Yes

*For this section a large eutherian carnivore is defined as a cat (*Felis catus*) or larger.

Chapter 8

Bait take and access to encapsulated toxicants by non-target species

Chapter overview

One technique currently being trialled to minimise the access of non-target species to the toxicant in baits is to encapsulate it within a hard shelled delivery vehicle (HSDV). In the preceding chapter I conducted a desktop analysis to determine which non-target species would be most likely able to access a toxicant encapsulated within a HSDV. To determine the validity of this analysis for a subset of the species assessed, I undertook a field experiment to determine more critically which non-target species would consume the Eradicat® cat bait and which of these would be able to access the encapsulated toxicant. Field testing was carried out in Far East Gippsland.

I used a non-toxic systemic marker dye, Rhodamine B (Rb) as a toxicant substitute. When ingested, Rhodamine B leaves a band within actively growing hair that is visible under a fluorescent microscope. I distributed Eradicat® baits that were injected directly with an aqueous solution of Rb at two sites. At a further two sites I distributed baits in which the Rb had been encapsulated within a HSDV that was then inserted into the baits. I subsequently trapped small and medium sized animals at each of these sites. Captured animals had mystacial vibrissae (whiskers) plucked from the snout and analysed for Rb banding.

Four of the five non-target species captured had banding in the vibrissae, indicating that they had consumed some bait material but not necessarily all of the bait. At the sites where the Rb was encapsulated within HSDVs, three of the five species had clear banding in vibrissae. This indicated that they would be able to access a toxicant enclosed within a HSDV.

Testing of the HSDVs revealed that they rapidly lose structural integrity within the Eradicat® bait media, with integrity being reduced significantly within an hour of insertion into the bait. The Rb began to leach from the HSDVs within 3 hours, and then increased with the time since insertion of the HSDV into the bait. Integrity testing ceased at 13 hours post-insertion as the seal at the ends of the HSDV had by then failed completely. Any pressure on the HSDV at that time caused liquefied Rb to be ejected from the ends of the HSDV into the bait media. By 48 hours post-insertion, the HSDV could not be removed from the bait without causing it to rupture completely.

Introduction

Invasive predators often impact negatively on agricultural production and on the conservation status of endangered native wildlife (Kinnear *et al.* 1988; Saunders *et al.* 1995; Twigg and King 1991). Controlling these predators can result in significant benefits for both endangered species conservation and for increased agricultural production (Kinnear *et al.* 1998; Murray *et al.* 2006) The primary method of controlling invasive predators in Australia is through the use of poison baiting (Algar and Burrows 2004; Olsen 1998; Saunders *et al.* 2010).

Despite the ubiquity of its use, control of invasive predators by poisoning has raised concerns regarding the consumption of baits and toxicants by non-target species and the impact that this may have on the population dynamics of these species (Calver *et al.* 1989a; Calver *et al.* 1989b; Jackson *et al.* 2007). The most common toxicant used to control invasive predators in Australia is 1080 (sodium monofluoroacetate). Fluoroacetate occurs naturally in many Australian native plants across the western and northern areas of Australia. Access to fluoroacetate has resulted in most native animals in these areas having a relatively high natural resistance to 1080 that is not present in the invasive species (Twigg and King 1989; Twigg and King 1991). This allows 1080 baits to be laid on the ground surface when targeting introduced animals; if bait is consumed by non-target species it is likely to have little deleterious effect. Native animals in the south-eastern areas of mainland Australia and Tasmania show significantly lower tolerance to 1080 than their northern and western conspecifics (Fleming *et al.* 2001; King *et al.* 1978). As a result, baits containing 1080 are commonly buried in these areas to minimise access by non-target species but still allow some predators, especially red foxes, to find them. While this is an important strategy to minimise access by non-target species to the toxicant (Dexter and Meek

1998; Saunders *et al.* 1995; Thomson and Kok 2002), it still does not preclude a range of native mammals from excavating the baits and consuming them (Fairbridge *et al.* 2003; Fairbridge *et al.* 2000). In addition, there is little indication that feral cats will dig up buried baits, making this method of bait presentation unsuitable for cat control.

Studies conducted in Western Australia and on French Island, Victoria have shown that feral cats will consume aerially-laid baits that have a wet meat base (Algar *et al.* 2002; Burrows *et al.* 2003; Johnston *et al.* 2011). Research has also been undertaken to assess the suitability of encapsulating the toxicant in a hard shelled delivery vehicle (HSDV) to minimise the likelihood of non-target species encountering the toxicant in baits laid on the ground surface (Hetherington *et al.* 2007; Marks *et al.* 2006; also see chapter 7 of this thesis).

Initial likelihood of bait-take by non-target species can be assessed using a decision tree process (as demonstrated in chapter 7). However, field and pen trials are still an important step in ascertaining if native animals can access the toxicant in baits if they have been assessed as potentially susceptible by the decision tree process. In this experiment, I examined the ability of five small and mid-sized native mammals to access toxicant encapsulated within a HSDV. Each of these was deemed capable of consuming the bait media in the desktop analysis. I used a non-toxic bait marker encapsulated in the HSDV and then inserted this into non-toxic Eradicat® baits.

Methods

Marker

Rhodamine B (IUPAC name: [9-(2-carboxyphenyl)-6-diethylanimo-3-xanthenylidene]-diethyammonium chloride; molecular formula: $C_{28}H_{31}ClN_2O_3$) was used as the marker dye for this experiment. Rhodamine B (Rb) acts as a systemic marker that produces a persistent, but harmless, mark when consumed by animals. It produces short term staining on the lining of the intestinal tract and noticeably pink-stained scats on passing. The mark also appears in claws and hairs that there actively growing at the time of ingestion (Fisher 1999; Johns and Pan 1981). All hairs have a cycle of growth interspersed with resting periods. Mystacial vibrissae have very short resting periods so are most likely to be actively growing at the time of ingestion of the marker dye. Rhodamine B banding in vibrissae appears as a bright band between 1 and 6 mm in length when viewed under a fluorescent microscope (Fisher *et al.* 1999).

For this experiment, Scientec Research Ltd (Melbourne, Victoria) encapsulated 25 mg of Rb powder in the same HSDV as being developed for use with actual toxicants. Additional Rb powder was also supplied by Scientec Research Ltd which was then weighed and mixed into an aqueous solution with a concentration of 25 mg ml^{-1} allowing it to be directly injected into the baits.

Bait

Non-toxic Eradicat® feral cat baits were sourced from the Department of Environment and Conservation (DEC) bait factory in Western Australia This bait is based on a moist meat-based medium consisting of kangaroo meat, chicken fat, digests and flavour enhancers (Patent number AU 781829) (Algar and Burrows 2004; Hetherington *et al.* 2007). Baits are shaped in the form of chipolata sausages, with each bait ~15 g in weight and 10 cm in length.

One hundred baits were chosen at random from those supplied by DEC and had a single HSDV containing Rb inserted into them at about the midpoint. A further 100 baits were randomly chosen and injected with the aqueous Rb solution to determine which species consumed the baits. To ensure that all baits (i.e. those containing the HSDV with Rb and those injected directly with Rb) had the same concentration of Rb, the directly-injected baits were injected with 1 ml of the Rb solution.

All baits were sprayed with Coopex® (Bayer Crop Science Pty Ltd, Hawthorne East, Victoria) mixed to the manufacturer’s specifications (25 g Coopex in 2.5 l water) and allowed to dry. This was intended to act as a deterrent to ants and other invertebrates that might otherwise attack the baits. Baits were then sweated in the sun until fats in the bait medium started to appear on the surface of the bait (Algar and Brazell 2008; Algar and Burrows 2004). Two sites in tall forest habitat in Far East Gippsland were chosen at random to receive baits with the inserted HSDV and two further sites received the baits with the directly-injected Rb solution. The selected sites encompassed areas that are likely to be targeted for feral cat control in the future, and hence represent areas where the potential impacts of control on non-target species are important to quantify. The minimum distance between sites was about 1 km to ensure independence of the sites. Baits were spread haphazardly across each site at a rate of about 25 baits hectare^{-1} (~ 50 baits site^{-1}). Trapping was undertaken four weeks after the baits were distributed to allow the fluorescent banding to grow out along the vibrissae away from the follicle. The bulbs and follicles of hairs have a natural fluorescence that may interfere or mask any Rb banding. Allowing the banding to grow out along the vibrissae decreases the possibility of it being confused with this naturally present fluorescence (Fisher 1998).

Small mammal trapping

Four trapping grids were established in Far East Gippsland, each containing 64 type A Elliott traps (330 × 100 × 100 mm) (Elliott Scientific Equipment, Upwey, Victoria) in an 8 x 8 grid with 20 m spacing between the traps; the area covered by each grid was ~ 1.96 ha. Traps were placed in the location judged to be most suitable for small mammals within 2 m of each 20 m point to maximise capture success (Stewart 1979; Tasker and Dickman 2002). A wad of dacron fibre was placed in each trap to provide bedding and warmth for captured animals. Each trap was also placed in a plastic bag to prevent ingress of water and dew. Traps were positioned in such a way as to be shaded from the sun and, if necessary, additional bark and leaf litter were added to ensure that traps remained shaded and to camouflage traps from casual observers. Trapping for small mammals was undertaken in November 2009, with trap checks undertaken as close to first light as possible over three consecutive mornings.

Mid-sized mammal trapping

Wire cage traps (600 × 200 × 200 mm) (Mascot Wireworks, Homebush West, New South Wales) were borrowed from the Cann River Parks Victoria office for mid-sized mammal trapping. 24 traps were placed at each site using three of the lines used for small mammal trapping. Cage traps were trigger operated and closed when an animal pulled on the bait at the rear of the trap. Several large handfuls of leaf litter were added to each cage trap to provide bedding and warmth for captured animals. Each trap was fitted with a custom-made heavy PVC cover to reduce the amount of rain or dew that could affect the captured animals. All four sites were trapped simultaneously for mid-sized mammals in May 2010. A second batch of Rb laced baits was deployed for the mid-sized mammal survey. Any Rb banding present in the vibrissae from the baits deployed while sampling small mammals would have no longer been detectable due to the time between the small and mid-sized mammal surveys.

Trapping methods common to both small and mid-sized mammals

Traps were baited with a mixture of peanut butter, rolled oats and honey (Catling and Burt 1994; Dickman *et al.* 1983) compressed into a ball of approximately 2 cm diameter and placed in the back of the trap. To minimise possible influences from differing proportions of ingredients, the bait was made using the same ‘recipe’ on each occasion (400 g honey, 1 100 g rolled oats and 1 600 g smooth peanut butter – Buckmaster (2005)).

Traps were checked from first light each morning over three consecutive mornings with captured animals identified, weighed and sexed. Each animal was fitted with an individually numbered lightweight aluminium (< 0.18 g) self-piercing fish tag (Model 1005-1, National Band and Tag Co. Newport, Kentucky) to enable recaptured animals to be identified. The ear was swabbed with Betadine® solution (Sanofi-Aventis Pty Ltd, Rydalmere, New South Wales) prior to and following the fitting of the tag to minimise the risk of infection (Sharp *et al.* 2007). The tags were fitted using the proprietary applicator.

Traps in which animals had been captured were removed and replaced with a clean trap and bag. Dirty traps and bags were washed in a disinfectant solution, rinsed thoroughly and allowed to air dry before being reused (Tasker and Dickman 2002). For the small mammal trapping, all traps were washed in disinfectant when being moved between sites to minimise the possibility of transferring soil borne pathogens between sites.

Vibrissae collection and analysis

Captured animals were transferred to a clean bag and four vibrissae were removed from each side of the snout using flat-bladed tweezers (Fairbridge *et al.* 2003; Fisher 1998; Fisher *et al.* 1999). Vibrissae were sealed in small paper bags (75 × 125 mm) (Prospectors Earth Sciences, Glenwood, New South Wales), stored in a dark, airtight container and then refrigerated to minimise the leaching of Rb bands. For analysis, the vibrissae were gently washed in water then ethanol and allowed to air dry to remove any particulates that may have been stuck to their surface. Four vibrissae were then mounted per slide using DPX, a proprietary non-fluorescing mountant. Once the mountant was dry, slides were inspected under a fluorescent microscope using an ultraviolet light source with a wavelength of 360 nm.

HSDV integrity in the bait media

Based on the initial results of the small mammal trapping, I decided to test the integrity of HSDVs in the baits over time. Ten baits were chosen randomly and fitted with HSDVs containing Rb. These baits were then sprayed with Coopex® and sweated as described above. After 24 hours these baits were broken open and the HSDVs inspected for signs of leakage and loss of structural integrity. Following this, a further 96 baits were chosen at random and fitted with a HSDV containing Rb. These baits were sprayed with Coopex and sweated, as above. Baits were then placed on the forest floor to replicate field conditions and covered with fine mesh (1 cm) chicken wire to prevent access by birds and other animals. The HSDVs were removed from four randomly chosen baits each hour. The baits were inspected for leakage within the bait and then tested for crush integrity using a custom made press (see below). Hourly sampling ceased when integrity testing resulted in Rb being forced out the ends of the HSDV without actually puncturing the polymer. At this stage, I assumed that any animal eating the bait could compress the HSDV sufficiently to eject the Rb into the bait media. Two further tests of HSDV integrity were attempted at 24 and 48 hours after insertion into the bait media.

HSDV integrity testing

A press was constructed to enable HSDVs to be compressed against a set of electronic scales until they either broke or leaked Rhodamine B (Figure 23). A threaded rod was used to exert pressure on an HSDV that was positioned in the press; this was lathed to a diameter of 2 mm and rounded slightly on the end to replicate the approximate size of a marsupial carnivore tooth. As shown in chapter 7, such carnivores are the most likely non-target species to be impacted in cat-control programs. Once the HSDV was placed on the scales, they were zeroed to ensure that only the weight used to crush the HSDV was recorded. The threaded rod was screwed down at a uniform rate until the HSDV either broke or leaked Rb, at which time the weight as shown by the scales was recorded. Testing was conducted using four randomly chosen HSDVs on each occasion to obtain an average weight at which the integrity of the HSDV was compromised.

Results

A total of 65 captures of small and mid-sized mammals was made from 840 trap nights. Trapping resulted in the capture of 39 individual small mammals from three species, as well as two long nosed bandicoots (*Perameles nasuta*) and a single long nosed potoroo (*Potorous tridactylus*) (Table 17). No dusky antechinus (*Antechinus swainsonii*) were captured at sites where baits contained HSDVs, and no long nosed potoroos were captured at sites containing baits with directly-injected Rb.

The overall capture rate tended to be higher at the sites with baits containing HSDVs but the difference was not significant (= 2.6, *P* ns). There was no difference in the number of unique individuals captured in each treatment (= 0.95, *P* ns).

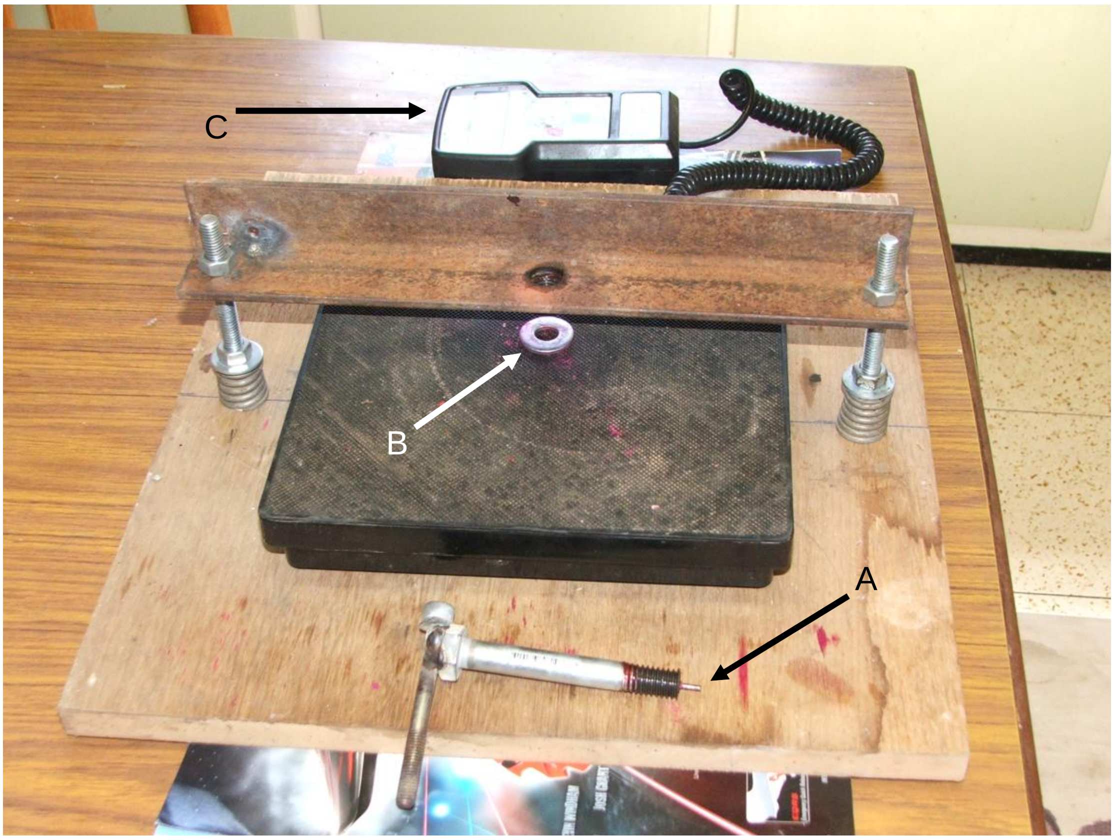

Figure 23. Press built for integrity testing of HSDV. A = bolt used to test integrity of HSDVs. For testing, this is screwed into the hole immediately above the metal washer. Note lathed end to replicate the shape of a tooth. B = metal washer used to stabilise HSDV during testing. C = electronic scales screen. Photograph: T. Buckmaster.

Table 17. The numbers of individual small and mid-sized mammals captured overall and in each of the two treatment areas, i.e. sites with baits containing HSDVs with encapsulated Rhodamine B dye, and sites with baits containing directly-injected Rhodamine B. Total captures inclusive of recaptures shown in parentheses. Recaptures are not shown for the two treatments.

Species	Captures	Bait direct injected	Bait containing HSDV
Bush rat (*Rattus fuscipes*)	29 (44)	12	17
Agile antechinus (*A. agilis*)	6 (12)	3	3
Dusky antechinus (*A. swainsonii*)	4 (4)	4	0
Long nosed bandicoot (*P. nasuta*)	2 (4)	1	1
Long nosed potoroo (*P. tridactylus*)	1(1)	0	1
total	42 (65)	20	22

Bait consumption

No Rb banding appeared in the vibrissae of the long nosed potoroo. The remaining four species all had Rb banding in the vibrissae, demonstrating that these species did consume Eradicat® bait (Figure 24). Several bush rats had up to three bands in each vibrissa, suggesting these individuals accessed baits on multiple occasions. More of the bush rats captured had consumed baits than had not consumed baits, but this was not significant (= 1.33, *P* ns).

HSDV

Three of the four species captured at sites with Rb encapsulated in the HSDVs had Rb banding in their vibrissae (Figure 24). At the sites with baits containing HSDVs, more bush rats accessed the Rb than those that did not but the difference again was not significant (= 0.594, *P* ns). The single long nosed bandicoot captured at those sites had accessed the encapsulated Rb, as had two of the three agile antechinus captured.

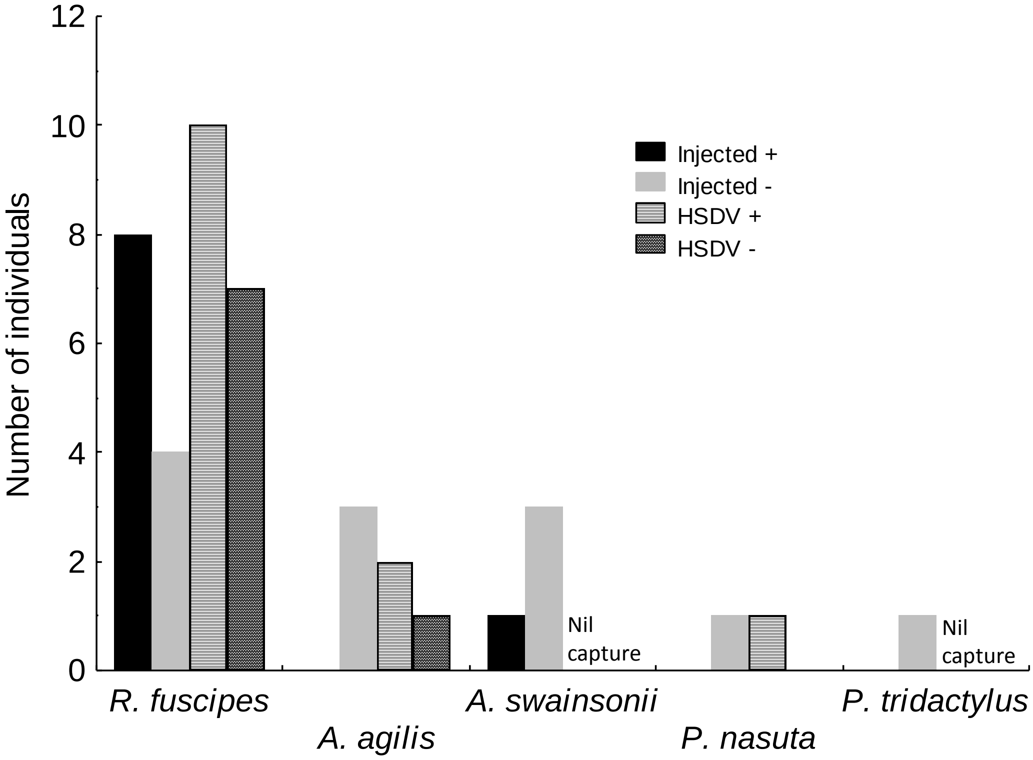

Figure 24. Numbers of each of five mammal species tested for Rb banding in each of the two bait treatments. Injected = Rb directly injected into the bait. HSDV = Rb encapsulated into a HSDV within the bait, + indicates Rb banding in vibrissae detected, - indicates Rb banding not detected.

HSDV crush integrity

The integrity of the HSDV fell significantly within the first hour of being placed within the bait media (Figure 25) ($t_{(6)} = 3.856$, $P = 0.008$). After 3 hours there were minor signs of Rb leaking from the ends of the HSDV where the polymer is folded to close off the end. After 5 hours there was substantial Rb staining within the baits around the folded ends of the HSDV. The amount of bait media stained with leaked Rb increased with each subsequent testing. After 7 hours the HSDV was very soft to the touch and care had to be exercised when removing it from the bait to prevent it splitting open as it was being removed. After 13 hours within the bait media, attempts to crush the HSDV resulted in liquefied Rb being expelled from the ends rather than the HSDV being punctured by the press.

After 24 hours in the bait media, the entire bait matrix surrounding the HSDV was stained with Rb, extending 4-5 mm in all directions (Figure 26). All the Rb powder in the HSDV had liquefied by this time. After 48 hours, the HSDVs had lost all structural integrity, and ruptured completely when being removed for integrity testing.

The polymer shell of the HSDV would shatter on failure when tested prior to being inserted into the bait at about 9.5 kg pressure in the integrity press. The shell similarly shattered on failure at the 1 h and 2 h tests albeit at a lower pressure (Figure 25). In testing at 3 hours and subsequently, HSDV failure resulted from the press puncturing the polymer shell rather than from it shattering. At 13 hours after insertion, testing forced liquefied Rb from the ends of the HSDV rather than puncturing or shattering the shell. Testing ceased at 13 hours (Figure 25).

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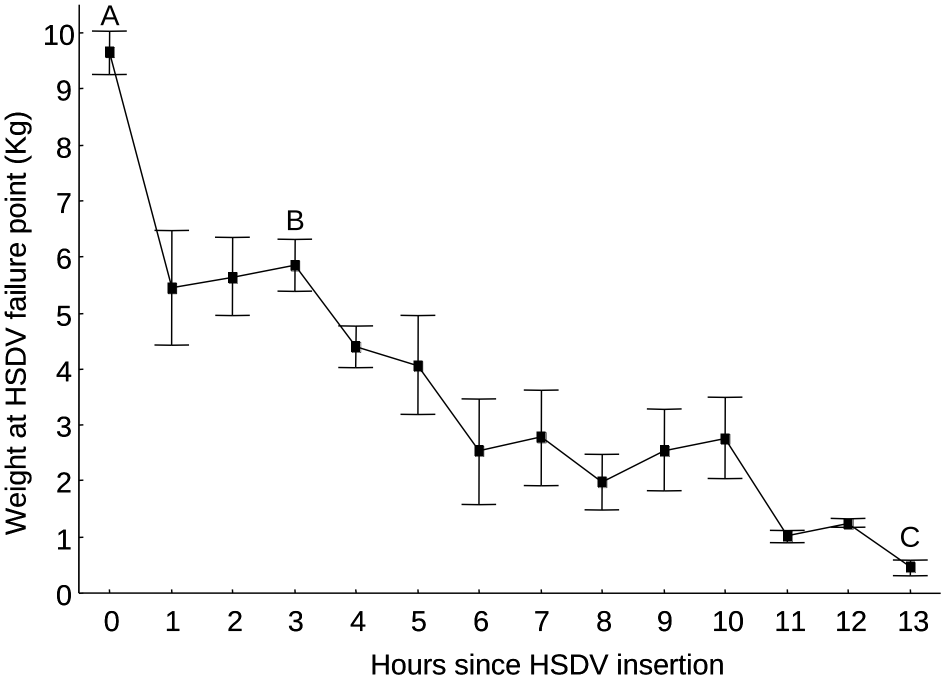

Figure 25. Results of hourly integrity testing of HSDV following insertion into bait media. A = pre insertion integrity. B = point where HSDV no longer shattered but was punctured by the integrity press. C = point where press forced Rb out the ends of HSDVs rather than puncturing the shell. Means are shown ± SE, with n = 4 for each hourly point.

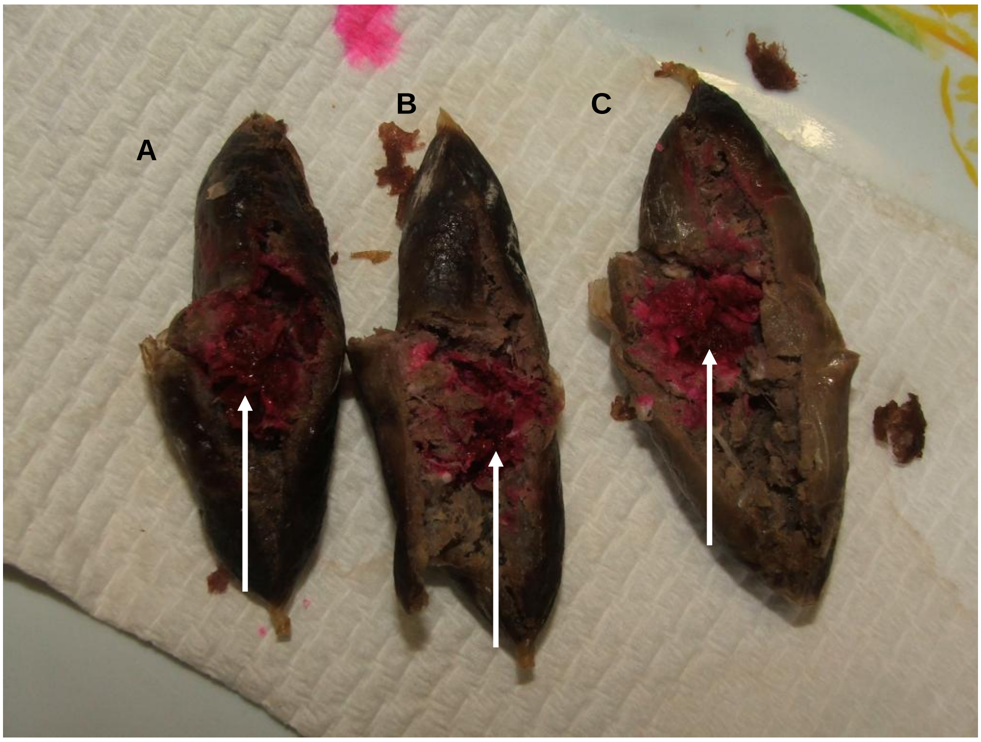

Figure 26. Eradicat® cat baits opened to remove HSDVs for testing. Pink staining is Rhodamine B that has leached out of HSDV. White arrows indicate location of HSDV prior to removal. A = 48 hours after insertion. B = 13 hours after insertion. C = 24 hours after insertion. Photograph: T. Buckmaster

Discussion

One of the primary concerns with laying toxic baits on the ground surface is the risk of them being consumed by non-target species. This has stimulated a considerable amount of research. For example, Fenner *et al.* (2009) tested non-target bait uptake by bush rats and brown antechinus (*Antechinus stuartii*) of wet meat baits used for wild dog control. These baits were injected with 6 mg of 1080 and 50 mg of Rb. After the laying of these baits, trapping revealed no animals with Rb banding in their vibrissae, leading the authors to conclude that no sublethal consumption of bait had occurred by either bush rats or brown antechinus. However, as the 1080 and the Rb were mixed together prior to injecting into the bait, any animal that consumed the Rb must also have consumed the 1080. No conclusion was made by Fenner *et al.* (2009) as to possible levels of lethal consumption. My research has revealed that both bush rats and the agile antechinus (recently split from the brown antechinus as a separate species by Dickman *et al.* (1998)) do indeed consume wet meat-style baits that have been laid on the ground. This makes them susceptible to non-target poisoning if the baits are directly injected with the toxicant.

The desktop analysis I conducted to determine which non-target species would consume baits used in a feral cat baiting program indicated that all five species captured during this experiment would be likely to do so (Chapter 7). Vibrissae from four of the five species contained Rhodamine B banding, indicating bait consumption. The single long nosed potoroo captured did not have banding in its vibrissae. However, it was a single animal; as this species has been found to consume baits on French Island (M. Johnston unpublished data), it is highly unlikely that this animal is representative of the population as a whole.

This experiment also demonstrated that it was possible for three of the five captured species to access the Rb encapsulated within the HSDV. Although the desktop analysis again indicated that the long nosed bandicoot and the bush rat were both capable of accessing toxicant within HSDVs based on previous experiments (e.g. Marks *et al.* 2006), it was not expected that the *Antechinus* species would be able to access the HSDV contents. This experiment demonstrated that the agile antechinus was capable of accessing the Rb within the HSDV. While no dusky antechinus were captured at the sites where the HSDV was employed, as they are larger than the agile antechinus, it is assumed that this species would also be able to access the contents of a HSDV when consuming bait.

The Rhodamine B banding in the vibrissae of those animals captured at the sites where the HSDV was deployed can be explained largely or entirely by the failure of the HSDV to maintain its integrity within the Eradicat® bait. Animals that consumed the bait media but rejected the HSDV would have been susceptible to consuming the Rb without having consumed the HSDV from a point just 3 hours after the HSDV was inserted into the bait. As the time within the bait media lengthened, the level of accessibility of the encapsulated Rb increased even if the HSDV was rejected by the animal consuming the bait. After 48 hours within the bait media, the HSDV had lost structural integrity to a point where gently trying to remove it from the bait resulted in it breaking open.

During this experiment, the baits were sweated for two hours prior to being taken to the field and distributed. Overall, it was about four hours from the time the HSDV was inserted into the bait media to the time they were distributed at the trapping sites. Thus any animal consuming the bait media from the time baits were distributed in the field was liable to have come in contact with the Rb that had leaked from the HSDV. Any animal consuming a bait more than 24 hours after the insertion of the HSDV would have scarcely noticed the presence of the HSDV within the bait as rupturing it took similar or lesser force as biting into or through the bait itself.

These results confirm that loss of structural integrity of the HSDV allowed the Rb to leach out into the bait media, or resulted in animals puncturing the HSDV while chewing the bait. In either case, animals that accessed the Rb showed banding of the vibrissae. In terms of toxicant delivery using the combination of HSDV and bait tested in this study, non-target animals consuming the bait would have been able to access the toxicant from just three hours after the insertion of the HSDV into the bait media. Due to the time taken to process the baits (inserting HSDV, spraying with

Coopex®, sweating and then deploying), access to the encapsulated toxicant would have been possible from the moment the baits were deployed in the field.

The point of failure of the HSDVs while in the bait was the end of the HSDV capsule where the polymer had been folded in during manufacture. Initial leakage of Rb was detected at the ends and, as more RB leached out, it diffused through the bait media adjacent to the HSDV. The liquefaction of the powdered Rb within the HSDV resulted from the moisture (water and melted fats following sweating) within the baits leaking into the HSDVs through the failed folds at either end.

Similar to the findings of this study, during preliminary field trials on Christmas Island it was noted that HSDVs lost integrity within the bait (Johnston *et al.* 2010a). The authors of this latter study suggested that this may have been the result of the baits being slightly acidic. Since the conclusion of the present study, the Curiosity® cat bait has now been buffered to a pH of 7.5 by Scientec Research Ltd to overcome the problem of the HSDV losing structural integrity within the bait media and allowing the animal access to the enclosed toxicant (Johnston *et al.* 2011; M. Johnston pers comm). These modified baits were used successfully in cat management operations in the arid environment of Dirk Hartog Island, Western Australia, in 2010 (Hilmer 2010; Johnston *et al.* 2010b). Further trials are now required to determine whether the modified bait medium will increase the longevity of the HSDVs under field conditions in other environments such as in the mesic forests of eastern Australia, or whether additional improvements are needed to make the concept viable generally under field conditions.

Acknowledgements

I sincerely thank Dave Algar from the Western Australian Department of Environment and Conservation for providing me with the baits used in this experiment. I also thank Michael O'Donaghue from Scientec Research Ltd for providing the Rhodamine B powder and HSDVs of encapsulated Rb. Thanks also to Mark Thornthwaite for assisting with designing and constructing the press used for integrity testing and for loaning me his electronic refrigerant-gas scales for the press.

Chapter 9

Synthesis

The primary aim of my research was to answer key questions about feral cats in tall forested ecosystems. These questions were a) how large are the home ranges of feral cats in tall forests and to what extent to their ranges overlap; b) how do these cats move through their environment and utilise their home ranges; and c) is there a bait suitable for delivering toxicants to feral cats and, if so, when is an optimal time to do so. While the focus of my work was restricted to the tall forests, many of the findings of the research should be broadly applicable to the management of feral cats and other predators regardless of habitat type or location. I briefly outline my key findings in the first part of this chapter, and then discuss their implications for management.

Knowledge of feral cat movement patterns and of how cats utilise areas within their home range can allow refinements to be made in the planning of management programs that actively exploit these behaviours, thus allowing more effective management of cats in many different environments. For example, baits can be set in places such as creek lines where it is most likely that cats will encounter them. Similarly, use of the decision tree process to determine which non-target species are at risk of accessing toxicants used during baiting campaigns can be adapted to the management of cats and many different pest species. This approach also allows comparisons to be made between management techniques to determine which is the most cost-effective for managing feral cats, yet still affords the greatest protection to non-target species.

The use of GPS telemetry on feral cats has enabled the gathering of accurate locational data in quantities previously not possible with conventional VHF telemetry, and this has in turn allowed patterns of intra-home range use by cats to be examined in detail. GPS technology provides further benefit in that, unlike conventional VHF telemetry, the researcher does not need to be in the field to obtain fix data. It is gathered automatically at the tasking rate programmed by the user into the GPS unit.

The feral cats in Far East Gippsland proved to be extremely elusive following their initial capture. As the GPS units stored all the fix data on board, the collars needed to be retrieved to allow the data to be gathered. Timed automatic collar release mechanisms were not readily available at the time this project commenced. The on-board VHF transponder allowed the location of cats to be readily identified and thus allowed trapping to be targeted to the appropriate locations. Despite very extensive efforts using several different trap sets and techniques over three years to recapture the feral cats with the GPS collars, only three were actually recaptured. Several others died, allowing the collars to be retrieved after the mortality sensor activated. I recommend that any future work being undertaken on feral cats should use GPS receivers fitted with timed automatic collar release mechanisms. This will allow the data stored on board the collar to be retrieved in an easier manner than having to recapture recalcitrant cats. Once the collar releases, the researcher can locate the collar using the inbuilt VHF transponder and thus readily retrieve it.

This was the first research undertaken on the home range of feral cats in tall forests in Australia. When compared with other studies of feral cat home ranges in Australia and New Zealand, feral cats in the tall forests of Gippsland were found to have home ranges smaller than those in alpine, arid and semi-arid areas but larger than those in modified grassland or farmland habitats. The size of feral cat home ranges appears to be governed generally by the availability of food resources. Cats in areas where there is a paucity of prey have larger home ranges than those in areas with abundant food resources. The high degree of overlap of female home ranges, including overlaps of core areas, shows that female cats are tolerant of other females and do not seek to actively exclude them. However, this overlap does not indicate that the cats are group living as can be found in localised areas of high food resources such as at rubbish tips.

The general convention when using Minimum Convex Polygon (MCP) analyses to determine home ranges is to use a MCP50 to define the core area of use. The large volumes of fix data gathered using the GPS collars in my research showed that the MCP50 definition of core area incorporated areas that the cats did not use as well as excluding areas of high cat use. As a result, continued use of this method for determining core areas is not recommended. By using instead the 50% isopleth of a

Kernel Density Estimator (KDE) the core area can be more accurately identified. Using the KDE50 also allows identification of multiple core areas within home ranges, rather than just a single area as with the MCP50 analysis.

The large volume of data gathered using GPS collars in this study also yielded insight into how animals move through their home ranges by analysing the step length and turn angles between successive fixes. These were analysed at three temporal intervals and revealed that cats employed a Lèvy walk style of movement. Using this pattern of movement allows cats to increase the probability of encountering prey items within their home range, especially as the predator is larger than its prey and the prey items are distributed sparsely across the landscape.

Modifying feral cat management programs to exploit such movement patterns should increase the probability of cats coming into contact with management devices, be these poison baits or traps. The poison bait delivery technique employed in Western Australia simulates sparsely distributed prey, in that clusters of baits are dropped at regular intervals from an aircraft (Johnston *et al.* 2011; Johnston *et al.* 2010b). While this results in a uniform program-wide distribution of baits at 50 / km^2, at a finer scale, the baits are distributed in relatively small areas with large areas of matrix with no baits.

All the cats for which GPS collars were recovered had large areas within their home ranges that they did not enter. These unused areas did not arise from the sampling regime of the collars as each had a degree of time slip between fixes that resulted in fixes being taken at all times throughout the day. My initial hypothesis that there was a deficiency of prey in the unused areas was not supported by the small and mid-sized mammal trapping data. The trapping showed that there was no difference in prey abundance between the habitats commonly used by the cats and those which they avoided.

Modelling of habitat parameters between the used and unused areas of cats' ranges revealed that elevation, vegetation community, slope of the land and the distance to the nearest stream were the most important factors in determining which areas were used and which were avoided. However, the global model was overdispersed ($\hat{c}$ =

6.33), indicating that other unmeasured parameters are involved in the determination of habitat use by cats. The recovery of a GPS collar that bore the teeth marks of a larger intraguild predator, either a fox or wild dog/dingo, suggested that the missing parameter in the model may be a predator avoidance strategy employed by cats in Far East Gippsland. It is possible that cats avoid areas within their home ranges to minimise their risk of predation from the larger predators that are present. Further support for this possibility comes from the analysis of GPS collar data from feral cats on islands where there are no larger intraguild predators present (Hilmer 2010; Johnston *et al.* 2010b). On these islands the large unused areas seen in the ranges of Gippsland forest cats are not present, and the home ranges of these insular feral cats tend to cover most or all of the areas encompassed within their range boundaries. I further discuss the implications for management of this apparent large predator avoidance strategy later in this chapter.

Feral cats are obligate carnivores and are often described as generalist predators (Bradshaw *et al.* 1996; Fitzgerald and Turner 2000), but some recent research suggests that individuals within populations can specialise in hunting particular types of prey (Dickman 2009). Feral cats in Far East Gippsland prey predominantly on the bush rat (*Rattus fuscipes*), which was also the most commonly caught prey species during my mammal trapping. Feral cats appear to avoid depredating agile antechinus (*Antechinus agilis*); this was the second most commonly captured small mammal species yet did not appear in any feral cat scats. The scansorial nature of agile antechinus may afford them a greater degree of protection from predation through being able to avoid capture by climbing trees; however the predominantly arboreal common ring-tailed possum (*Pseudocheirus peregrinus*) was present in about 18% of the cat scats. The agile antechinus had the lowest body mass of all the species captured during the mammal trapping and was also the smallest of the mammals in the cats’ diet. It is possible that cats do not prey on this species due to the lower energetic or nutritional return per prey item that it provides compared with larger species. However, previous research (Triggs *et al.* 1984) shows that cats will eat agile antechinus under certain conditions, perhaps when more profitable food sources are scarce.

The agile and dusky antechinus in Far East Gippsland have an earlier mating season than would be predicted based on more northern populations of these species. The subsequent death of all males in these species contributes to the decline in prey items for feral cats over winter. As a result, the times that cats are food stressed, and more liable to consume a bait during a poison baiting campaign, start and finish earlier in Far East Gippsland than would be predicted. Similarly, times when populations of non-target species are more vulnerable to non-target poisoning, such as when there are young in females' pouches, also occur earlier in Far East Gippsland than in other locations. The implication of this for the management of feral cats is discussed later in this chapter.

Bait take by non-target species in any poison baiting campaign is one of the primary concerns for land managers. Much assessment of bait-take potential by non-target species is undertaken at a local level and with limited consistency between sites. I employed a decision tree process to determine which Australian vertebrate animals would be most likely susceptible to accessing toxicants that were either injected directly into baits or encapsulated within a hard shelled delivery vehicle (HSDV). The Curiosity® cat bait, which utilises the toxicant para-aminopropiophenone (PAPP) encapsulated within a HSDV, was used as the basis for my analyses. The decision tree process allowed those animals that are likely to be susceptible to accessing toxicants to be readily identified and listed for further testing through field and pen trials. Significantly fewer non-target species were assessed as being able to access toxicant when it is encapsulated within a HSDV compared with when it is injected directly into bait media.

To test the validity of the desktop analysis, the marker dye Rhodamine B (Rb) was injected into Eradicat® baits that were then deployed at two sites in Far East Gippsland. Baits containing Rhodamine B encapsulated within HSDVs were distributed at two further sites. Vibrissae taken from small and mid-sized mammals captured at these sites indicated that four of the five species captured consumed the baits and would therefore be able to access any toxicant that had been directly injected into the baits. Analysis of the vibrissae also revealed that three of the five species had accessed the Rb even when it had been encapsulated within a HSDV. Subsequent testing of the HSDVs indicated that they rapidly lost integrity after being inserted into the baits. HSDV integrity declined continually until a point 13 hours after insertion when complete structural integrity was lost and any external pressure resulted in the encapsulated Rb being ejected into the surrounding bait media.

Implications of my findings for managing feral cats for conservation

Far East Gippsland

The high degree of overlap found in the home ranges and core areas of the feral cats in Far East Gippsland indicates that cat density in the region is higher than would be expected if the cats excluded others from their home ranges and core areas. It also indicates that the abundance of available resources for feral cats is high as they do not need to maintain exclusive areas to ensure adequate access to obligate resources such as food and shelter.

It also appears that the presence of larger intraguild predators could be influencing the behaviour of feral cats in the region. The presence of larger predators has been suggested or demonstrated in other areas to exert a level of suppression on the behaviour of cats (e.g. Glen *et al.* 2007a; Johnson *et al.* 2007; Molsher 1997; Molsher 1999). There are limited techniques available for the management of feral cats in the forests of Far East Gippsland. No toxicant is yet registered and able to be used in baiting programs for feral cats in Victoria. Similarly, legislative changes to the Victorian *Prevention of Cruelty to Animals Regulations 2008* now preclude the use of soft-jaw traps to catch feral cats in state forests. If feral cats are to be captured, cage traps must therefore be used, despite their ineffectiveness in forest habitats. As a result, management of feral cats may need to be undertaken by adopting different management practices such as the use of higher order intraguild predators. Baiting of canids, primarily red foxes, is currently undertaken over large areas of forest. It is recommended that a predator management experiment be undertaken in Far East Gippsland, similar to that of Risbey *et al.* (2000), to determine what role, if any, larger intraguild predators play in regulating the behaviour of feral cats. If such an experiment can be undertaken, it should also determine the resultant impacts on prey species. The results of such an experiment could then be used to guide how not only feral cats, but also the larger intraguild predators, should be managed.

When fully implemented, the Southern Ark project will be poison-baiting foxes over about 1 million hectares in Far East Gippsland. Fox abundance is being reduced as a result of the baiting by Southern Ark (Diment 2010). If the larger intraguild predators are exerting a suppressive influence on feral cats, the reduction in abundance of foxes (and possibly dingoes or wild dogs also) may result in mesopredator release, with cats increasing in activity and abundance. If this occurs, there is potential for there to be greater impact on the native mammals than if fox management did not occur (Risbey *et al.* 2000; see also Chapter 2). It is therefore recommended that regular small mammal surveying occur in conjunction with the trapping already being undertaken by Southern Ark for mid-sized indicator species. Together with the long-term regular sand-plot monitoring already being undertaken, this will indicate if feral cats are released from suppression following the reduction in fox abundance and what impact this has on the smaller prey species that are not currently being monitored by Southern Ark.

The findings of my small mammal trapping indicated that *Antechinus* species in Far East Gippsland breed earlier than would be expected based on photoperiodic triggers. This has implications for the control of predators as the most appropriate timing of management interventions for feral cats is when they are food stressed by reduced abundances of their prey. The optimal time for poison baiting feral cats in Far East Gippsland would be between the late August and mid November. At this time the *Antechinus* populations are low as a result of the male die off following the mating season. As this time is also when female antechinus are likely to be food stressed, surface-laid baiting should only occur during this time using either a HSDV that maintains structural integrity within the bait matrix or through suspending the baits beyond the reach of the female antechinus. Similarly, as foxes include *Antechinus* species in their diet (Diment 2010; Triggs *et al.* 1984), these findings could be used by the Southern Ark project to target times when foxes are more food stressed.

Trapping for small mammals in Far East Gippsland should avoid times when *Antechinus* species have pouch young due to the potential for the female to eject these young through stress when captured in a trap. Similarly, when the young have been deposited into a nest, they are still completely reliant on the mother for nourishment. Trapping at these times has the potential to negatively impact on these young through the female being restrained within a trap for extended periods.

Although the sample sizes obtained during my trapping were small, the short breeding season of *Antechinus* species is usually synchronous across local populations within a region (McAllan *et al.* 2006), so it is likely that the animals captured in my work reflected the life history timing of the broader species' populations. However, it is recommended that further trapping be undertaken in Far East Gippsland to definitively establish the temporal variation in the breeding season compared with the rest of the state. In chapter 6, I also raised the possibility that a remnant population of brown antechinus (*Antechinus stuartii*) may be present in the southern areas of Coopracambra National Park. A similar remnant population of *A. stuartii* or an equivalent taxon exists at nearby Mallacoota (McAllan and Dickman 1986). It is recommended that further sampling of *Antechinus* spp. be undertaken in Coopracambra National Park to determine if a remnant population of the brown antechinus does exist in that area, or whether other aberrant populations within this diverse and ubiquitous genus occur there.

Broader applications of my finding

Adapting management practices to better target pest species with control measures, or to conserve rare species, is not new (Olsen 1998). Techniques are being developed, or adapted from other fields of science, to describe the movement patterns and habitat selection processes of a number of species (for examples, see Robley and Gormley 2010). In this instance, the Lévy walk principle was originally developed to describe the movement of a particle in a fluid yet has now been found to describe the movement patterns of many animals as they move through their environment (Humphries *et al.* 2010; Reynolds *et al.* 2007b). While it is beneficial to have this knowledge, determining how to apply this newfound knowledge to the management of a species is of far greater benefit and yet very few published studies take this next step. To avoid this deficiency, suggestions are made on how management plans could be adapted to incorporate the knowledge gained from the research I have undertaken.

My research has shown that feral cats utilise a Lévy walk style of movement through their environment at all the temporal scales that I sampled. This type of movement pattern increases the probability that they will detect sparsely distributed prey items. Similarly, feral cats probably avoid certain areas within their home ranges when in sympatry with larger intraguild predators. This knowledge can be used to adapt feral cat management programs to target them more effectively and, potentially, increase the efficacy of the program. For example, where feral cats are in sympatry with larger intraguild predators, baits or traps should be deployed along creek lines or areas of greater structural complexity and thus avoiding areas that are potentially not used by the cats. Baits should also be deployed in a clumped manner rather than uniformly distributed (for examples of clumped baiting see Johnston *et al.* 2011).

The desktop analysis I developed will enable land managers Australia-wide to quickly assess which non-target animals in their region are potentially susceptible to non-target poisoning when using the Curiosity® feral cat bait as part of a management program. By modifying the assessment criteria in the analysis, the decision tree process that I used can be adapted for use with other pest animals. Similarly it can be adapted for use with different toxicants and bait combinations by modification of the assessment criteria. The decision tree process has also been used to identify potential sites of high feral cat impact across Australia and prioritise these sites for cat control (Dickman *et al.* 2010).

My experiment using the Eradicat® bait resulted in HSDVs rapidly losing their structural integrity. This allowed any non-target species that consumed the bait to come in contact with the encapsulated Rhodamine B. A similar loss of the structural integrity of HSDVs was observed by Johnston *et al.* (2010a) when also using them in conjunction with the Eradicat® bait. Following those observations, and since this experiment concluded, there have been modifications to the buffering of the pH of the baits, Although these should result in strengthening the integrity of the HSDVs, further experiments are now needed to determine whether the capsules will retain sufficient integrity for field use. The desktop analysis I undertook showed that the use of HSDVs to encapsulate toxicant within feral cat baits should significantly reduce the number of non-target species that can access the toxicant. The use of HSDVs in feral cat baiting programs will enable such programs to be used in the eastern states of Australia where the use of directly-injected 1080 cannot be used in surface laid baits due to concerns about poisoning of non-target species.

Conclusions

The research I have conducted and presented in this thesis should allow a more targeted approach to feral cat management while minimising the risks to non-target species. The analytical techniques that I used and applied to feral cats in Far East Gippsland are not restricted to that species or that region alone. They are applicable to a broad range of management situations involving the management of one species and the potential flow-on effects that the management intervention may have on other species. Similarly, the research into the behaviour of pest animals, how they move through their ranges and why they use or don’t use parts of their ranges, can be used to adapt management techniques in order to increase their effectiveness. For example, any pest animal that employs a Lévy walk pattern of movement should be especially susceptible to a baiting program in which baits are deployed in a sparse or patchy manner.

Of course, my research is by no means the final word on feral cats in the tall forests of Far East Gippsland. While it has provided hitherto unknown insights into the ecology and behaviour of this elusive predator, it should be seen as a starting point rather than as the end point for more detailed investigations elsewhere in the tall forest environment.

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Appendix 1

Diet of feral cats (*Felis catus*) in Far East Gippsland, Victoria

Introduction

The feral cat (*Felis catus*) was introduced to Australia with European settlement in the late 1700s (Abbott 2002). Since that time it has spread across the Australian mainland, to Tasmania and many offshore islands. It has been implicated in the decline of many native species, particularly those in the arid and semi-arid regions and on offshore islands (Burbidge and Manly 2002; Burbidge and McKenzie 1989; Dickman *et al.* 1993; Short and Smith 1994). The introduction of the European rabbit (*Oryctolagus cuniculus*) into Australia facilitated the spread of the feral cat by providing a readily available food resource. So too did the deliberate capture, breeding and release of cats into the wild in a misguided attempt at controlling the spread of the rabbit (Rolls 1969).

In most regions of Australia where the rabbit is present, it forms the primary food source for feral cats. Cats will continue to preferentially depredate rabbits even when rabbit populations have been reduced by some 90% (Molsher *et al.* 1999). This indicates that cats selectively depredate certain prey items, and switch to alternative prey species only when populations of the preferred prey have declined to a significant extent. If rabbits are uncommon, their prevalence in the diet of feral cats is reduced and the bulk of the diet then consists usually of native small mammals (Jones and Coman 1981).

The analysis of predator scats has been used previously to study predator diets (e.g. Kirkwood *et al.* 2005; Risbey *et al.* 1999; Saunders *et al.* 2004), predator prey dynamics (e.g. Mahon 1999) and to determine the presence of prey species within areas (e.g. Brunner *et al.* 1976; Friend 1978). Similarly, it can alert land managers to the presence of species not previously detected in those areas. For example, broad-toothed rat (*Mastacomys fuscus*) remains in a fox scat confirmed the presence of this species in a region where it had previously been known only from fossil records

(Diment 2010, Appendix 2) In addition, diet analysis has been used to show differential predation between size classes and sexes within single species of prey (Dickman *et al.* 1991).

In this appendix, I utilised cat scats collected in Far East Gippsland to determine the composition of prey species in the diet of feral cats of that region. I then compared this information with the only previous study on cat diet in the region and with the species captured during small mammal trapping undertaken in chapters 7 and 8 of this thesis.

Methods

Predator scats were collected by fellow PhD student, Alex Diment between January 2008 and August 2009 while undertaking regular scat transects in four of the monitoring areas within the Southern Ark project (see Diment 2010, chapter 7, p 157) (Figure 27). Scats were air dried and then the surface was scraped using a sterile razorblade to remove epithelial cells. Scrapings were forwarded to the Wildlife Forensic Laboratory at the University of Western Australia to confirm the specific identity of the predator that excreted the scat (Diment 2010) using melt-curve analyses (Berry and Sarre 2007). Of the 697 scats collected by Diment (2010), 22 were identified as originating from a cat.

The 22 cat scats were forwarded to hair identification expert, Barbara Triggs (Dead Finish, Genoa, Victoria) for content analysis. The hair, bones and teeth of prey items consumed by cats are indigestible and are passed with other faecal matter following digestion of the balance of the animal (Brunner and Coman 1974; Triggs 1996). The morphometric features of the recovered guard hairs such as size, medulla pattern, cross section shape and scale pattern, allow the identity of the prey item to be determined with a high degree of accuracy for most species (Brunner and Triggs 2002; Lobert *et al.* 2001).

Insufficient scats were obtained to allow comprehensive dietary analyses to be undertaken. The percentage occurrence of each prey species was determined and compared with that found by Triggs *et al.* (1984), the only other study of feral cat diet in the region.

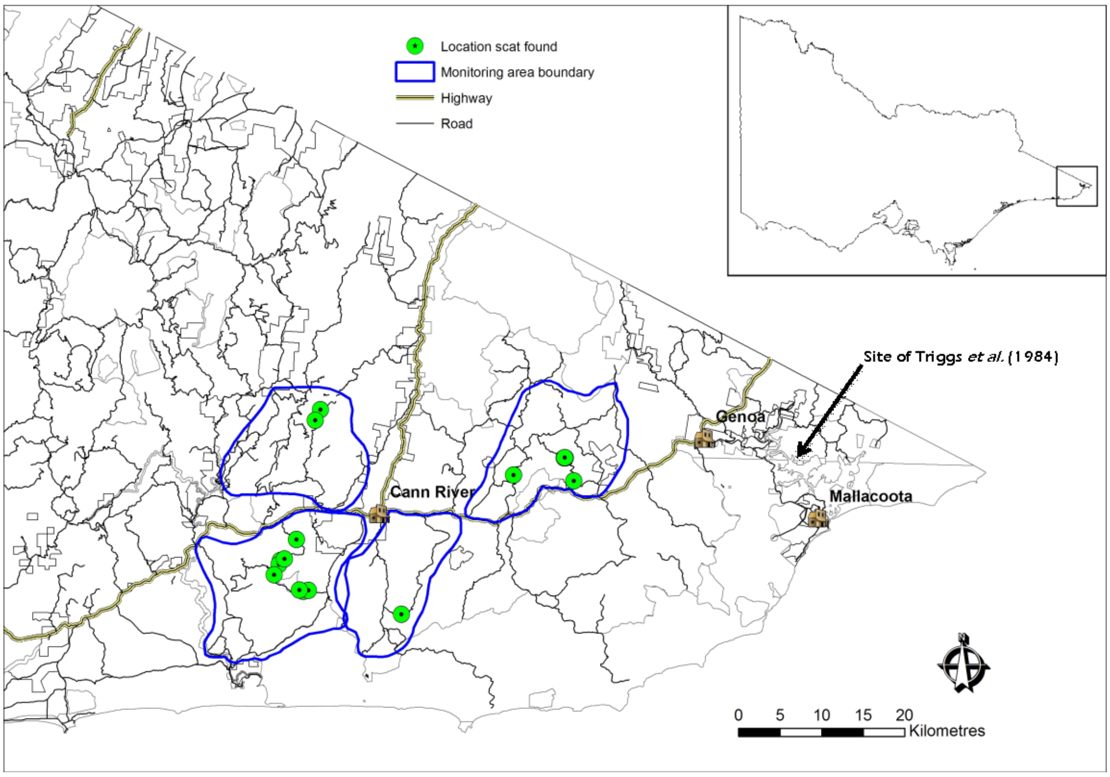

Figure 27. Locations of 22 feral cat scats collected by A. Diment while undertaking regular scat collection transects. Note: some scats were found adjacent to each other so only one marker may be visible for multiple scats. Background data were obtained from geospatial layers provided by the Department of Sustainability and Environment, Victoria.

Results

A total of 697 predator scats were collected by Diment (2010) during his study on fox diet. Only 22 (3.15%) of these were identified as belonging to cats. The hairs of seven mammal species (eight including *F. catus*) were detected in the analysed scats (Figure 28). A single scat contained no mammalian remains. Bird feathers and reptile scales were detected in only one scat. Bush rats (*Rattus fuscipes*) were the most common prey species detected in the scats. One scat also contained hair from *Trichosurus* sp., probably *T. vulpecular*. This genus is difficult to separate into individual species through hair analysis (Lobert *et al.* 2001) and is left at the species level for this analysis. Agile antechinus were not detected in any of the scats analysed for this study yet were found in 15% of the feral cat scats (n = 56) analysed by Triggs *et al.* (1984).

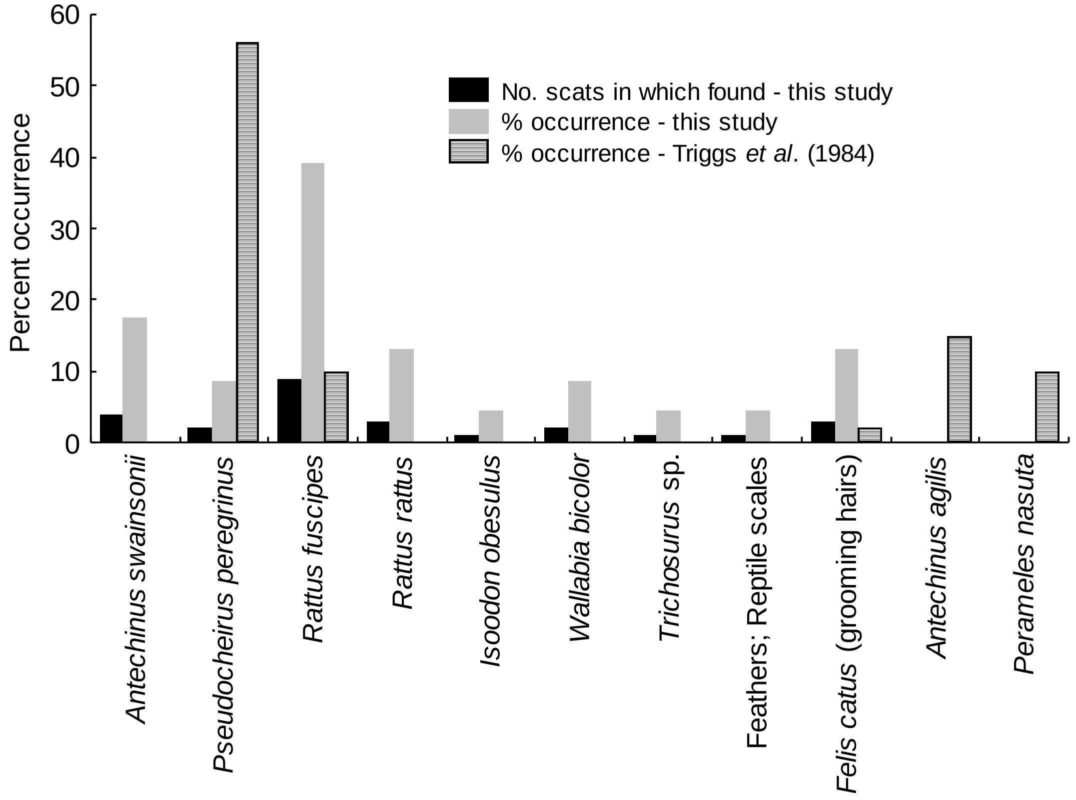

Figure 28. Percentage occurrence of dietary items in the feral cat scats found by A. Diment in Far East Gippsland (light grey bars). Also shown is the percentage occurrence of dietary items found by Triggs *et al.* (1984) (horizontally banded bars). The number of scats in which each species was detected in this study is also shown.

Discussion

Mammals are the dominant prey items for feral cats in Far East Gippsland occurring in about 95% of all scats collected in the present study and in the bulk of those analysed by Triggs *et al.* (1984). Native mammals formed the bulk of the mammalian component of the diet with the introduced black rat (*R. rattus*) being detected in only 13% of the scats. The bush rat (*R. fuscipes*) was the most common native animal detected in the cat scats. Small mammal trapping (see chapters 5 and 6) revealed that this was also the most commonly caught species in Far East Gippsland. The proportion of bush rats in the diet of cats was higher than that found by Triggs *et al.*

(1984) who found this species in only 10% of the scats they analysed. Conversely the proportion of scats containing common ring-tailed possum (*Pseudocheirus peregrinus*) hair was much lower in this study than that by Triggs *et al.* (1984) who found this species to be the most common prey item for cats (Figure 28).

There appears to be a marked under-representation of the agile antechinus in the diet of the feral cat as found in this study. Triggs *et al.* (1984) found agile antechinus (recorded then as *Antechinus stuartii*) remains in 15% of all the cat scats analysed. Similarly, hairs of agile antechinus were found in over 25% of feral cats stomachs analysed from Victoria's eastern highlands (Coman and Brunner 1972; Jones and Coman 1981). Agile antechinus were trapped in greater numbers than dusky antechinus at all the sites used for small mammal trapping in my study, yet dusky antechinus were detected in about 18% of the scats. One possible reason for the paucity of agile antechinus in the diet of cats is the lower body mass of this species compared with that of bush rats and dusky antechinus. The lower body mass would result in less energetic return for cats for each successful capture than it would for the larger animals, possibly rendering the agile antechinus energetically unprofitable to hunt when in sympatry with the larger prey items. If this is correct, it remains uncertain why agile antechinus occurred in the diet of feral cats in the study of Triggs *et al*. (1984). It is possible that cats were more nutritionally stressed in the earlier study and thus more likely to take any available prey than in my study. The early 1980s were characterised by regional droughts that could have reduced the general availability of prey for feral cats, but this possibility requires testing with more extensive data.

The presence of swamp wallaby (*Wallabia bicolor*) in cat scats may be indicative of scavenging on the part of the cat. Adult swamp wallabies are far larger than feral cats and it is highly unlikely that a cat could subdue and kill an adult (see Menkhorst and Knight (2001) for a full description of both species). Alternatively, it is possible that cats depredated juveniles, which are much smaller than adults and would be more readily able to be ambushed, subdued and killed. Feral cats are known to depredate adult rufous hare-wallabies (*Lagorchestes hirsutus*) (Gibson *et al.* 1995; Gibson *et al.* 1994) which are similar in size to juvenile swamp wallabies.

Birds and reptiles formed only a small proportion of the diet of feral cats in Far East Gippsland and were found in about 4% of scats. This is similar to the findings of other studies undertaken in Victoria (e.g. Coman and Brunner 1972; Triggs *et al.* 1984) but is lower than that found in other regions of Australia where birds and reptiles contribute a much larger proportion of the diet of feral cats. For example, in central Australia, birds and reptiles may occur in nearly 70% of all cat scats (Paltridge 2002), while in mixed forest habitats in New South Wales Glen *et al.* (2011) found bird remains in over 30% of cat scats.

The differences in prey proportions in the diet of feral cats between this study and that of Triggs *et al.* (1984) may be due to the differing locations of the studies, with this study being undertaken over a larger area some 40 kilometres west of that studied by Triggs *et al.* (1984). That study was also confined to the system of tracks and trails to the east of the Mallacoota inlet. It is possible that there is a difference in species composition between the two areas; however, the level of prey availability in that study and across the entire area of this study is unknown.

Care must be taken not to assume that feral cats are having a negative impact on prey species simply on the basis of including them in the diet. It does not indicate that feral cats are having a detrimental impact on the species at a population level that in turn leads to a decline of the prey species (Denny and Dickman 2010). Impact at a population level will occur only when the level of harvest by cats exceeds the ability of the prey population to replace what is consumed. For example, foxes (*Vulpes vulpes*) regularly depredate bush rats (Diment 2010; Friend 1978; Saunders *et al.* 2004) yet do not appear to have an appreciable negative impact at the population level. Banks (1999) determined that foxes take only what is termed the “doomed surplus” or that portion of the population that was not likely to survive even in the absence of fox predation.

The European rabbit (*Oryctolagus cuniculus*) has been found to be the preferred prey item of feral cats in many regions of Australia, including other parts of Victoria (e.g. Catling 1988; Coman and Brunner 1972; Molsher *et al.* 1999); however, this is not the case in Far East Gippsland. The remains of rabbits were not detected in any of the scats analysed in this study nor in that of Triggs *et al.* (1984). Rabbits are present in the Far East Gippsland region and were seen regularly both on and adjacent to the cleared land in the region and along forest roads substantial distances from cleared areas over the period that the scats were collected (T. Buckmaster, pers. observation). Rabbits are consumed, albeit in low numbers, by both foxes and wild dogs/dingoes (*Canis familiaris* / *Canis lupus dingo*) in the region (Diment 2010; Friend 1978; Triggs *et al.* 1984). In other nearby regions, rabbits are consumed in larger number by cats (Coman and Brunner 1972; Jones and Coman 1981). The substantial variation in diet between cat populations in different areas is most likely due to a combination of previous feeding experiences, possibly to the degree of hunger, and to prey availability (Bradshaw *et al.* 2000).

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Appendix 2

Potential for ingestion by non-target Australian animals of Curiosity$^{®}$ baits with toxicant enclosed in a hard shell delivery vehicle – Electronic supplement

A CD ROM containing the complete assessment for all species is attached to this page.

Note: The contents of this disc were scanned for potential viruses and other materials that may be harmful to your computer prior to being included on the disc. None were found during that scan. **However, I strongly recommend that you scan the file on this disc using your own anti-virus and anti-malware software prior to opening it.**