

# IMPROVING MAMMALIAN REINTRODUCTION SUCCESS IN THE AUSTRALIAN ARID ZONE



**Katherine Elizabeth Moseby**

School of Earth and Environmental Science, Faculty of Science

The University of Adelaide

Thesis submitted for the degree

Doctor of Philosophy

June 2012

## Thesis Declaration

This work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution to Katherine Moseby and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. I give consent to this copy of my thesis when deposited in the University Library, being made available for loan and photocopying, subject to the provisions of the Copyright Act 1968. The author acknowledges that copyright of published works contained within this thesis (as listed below) resides with the copyright holder(s) of those works. I also give permission for the digital version of my thesis to be made available on the web, via the University's digital research repository, the Library catalogue, the Australasian Digital Theses Program (ADTP) and also through web search engines, unless permission has been granted by the University to restrict access for a period of time.

Moseby, K.E.<sup>1,2</sup>, Read, J.L.<sup>1,2</sup>, Paton, D.C.<sup>2</sup>, Copley, P.<sup>3</sup>, Hill, B.M.<sup>4</sup> and Crisp, H.M.<sup>1</sup> (2011). Predation determines the outcome of 11 reintroduction attempts in arid Australia. *Biological Conservation* **144**, 2863-2872.

Moseby, K.E.<sup>1,2</sup> and Hill, B.M.<sup>4</sup> (2011). The use of poison baits to control feral cats and red foxes in arid South Australia 1. Aerial Baiting Trials. *Wildlife Research*, **38**, 338-349.

Moseby, K.E.<sup>1,2</sup>, Read, J.L.<sup>1,2</sup>, Galbraith, B.<sup>1</sup>, Munro, N.<sup>5</sup>, Newport, J.<sup>5</sup> and Hill, B.M.<sup>4</sup> (2011). The use of poison baits to control feral cats and red foxes in arid South Australia II. Bait type, placement, lures and non-target uptake. *Wildlife Research* **38**, 350-358.

Moseby, K.E.<sup>1,2</sup>, Neilly, H.<sup>6</sup>, Read, J.L.<sup>1,2</sup> and Crisp, H.A.<sup>1</sup> (2012). Interactions between a top order predator and exotic mesopredators. *International Journal of Ecology* Article ID 250352, doi:10.1155/2012/250352.

Moseby, K.E.<sup>1,2</sup>, Cameron, A.<sup>7</sup> and Crisp, H.A.<sup>1</sup> (2012). Can predator avoidance training improve reintroduction outcomes for the Bilby (*Macrotis lagotis*) in arid Australia? *Animal Behaviour* **83**(4) 1011-1021.

<sup>1</sup> Arid Recovery, P.O. Box 147, Roxby Downs, Australia, 5725.

<sup>2</sup> The University of Adelaide, North Terrace, Adelaide, Australia, 5005.

<sup>3</sup> The South Australian Department for Environment and Natural Resources, G.P.O. Box 1047, Adelaide, Australia, 5000.

<sup>4</sup> The Northern Territory Department of Natural Resources, Environment, the Arts and Sport, P.O.Box 496, Palmerston, Darwin, Australia, 0831.

<sup>5</sup> Fenner School of Environment, The Australian National University, Canberra, Australia, 0200

<sup>6</sup> 118 Utah Drive, Moranbah, Queensland, Australia, 4744

<sup>7</sup> Queensland Department of Environment and Heritage Protection, Kabool Road, West Burleigh Queensland, Australia, 4221

Signed \_\_\_\_\_

# TABLE OF CONTENTS

<b>1. CHAPTER ONE : Literature review and contextual statement</b>	<b>8</b>
1.1 <i>Mammalian extinctions in arid Australia</i>	8
1.2 <i>Mammalian reintroductions</i>	11
1.2.1 Criteria for success of mammalian reintroductions	12
1.2.2 Intrinsic factors affecting reintroduction success	13
1.2.3 Extrinsic factors affecting reintroduction success	16
1.2.4 Post-release monitoring	18
1.3 <i>Contextual statement</i>	18
1.3.1 The influence of predation	20
1.3.2 Improving reintroduction protocols	21
1.3.3 Predictive tools for post-release monitoring	23
1.3.4 Conclusion	23
1.4 <i>Study Area</i>	24
1.4.1 Arid Recovery Reserve	24
1.4.2 Wild West Zone	29
1.4.3 Dingo Pen	30
1.5 <i>Study Species</i>	30
1.5.1 Greater stick-nest rat ( <i>Leporillus conditor</i> )	30
1.5.2 Greater bilby ( <i>Macrotis lagotis</i> )	31
1.5.3 Burrowing bettong ( <i>Bettongia lesueur</i> )	32
1.5.4 Western barred bandicoot ( <i>Perameles bougainville</i> )	33
1.5.5 Feral cat ( <i>Felis catus</i> ) red fox ( <i>Vulpes vulpes</i> ) and European rabbit ( <i>Oryctolagus cuniculus</i> )	34
<b>2.CHAPTER TWO: Predation determines the outcome of 10 reintroduction attempts in arid South Australia.</b>	<b>36</b>
2.1 <i>Introduction</i>	38
2.2 <i>Methods</i>	40
2.2.1 Study sites	40
2.2.2 Reintroductions	42
2.2.3 Monitoring	43
2.2.4 Success criteria	45
2.3 <i>Results</i>	48
2.3.1 Main Enclosure reintroductions	48
2.3.2 Red Lake reintroduction	50
2.3.3 Wild West reintroductions	50
2.4 <i>Discussion</i>	52
2.5 <i>Acknowledgements</i>	55
2.6 <i>References</i>	56
<b>3. CHAPTER THREE : The use of poison baits to control feral cats and red foxes in arid South Australia.</b>	
<b>I. Aerial baiting trials.</b>	<b>61</b>

3.1 Introduction	63
3.2 Methods	65
3.2.1 Study Area	65
3.2.2 Poison Baiting	65
3.2.3 Change in activity	67
3.2.4 Data Analysis	70
3.3 Results	71
3.3.1 Rainfall	71
3.3.2 Cats	71
3.3.3 Foxes	74
3.3.4 Rabbits	76
3.4 Discussion	79
3.5 Conclusion	81
3.6 Acknowledgements	82
3.7 References	82
<b>4. CHAPTER FOUR : The use of poison baits to control feral cats and red foxes in arid South Australia.</b>	
<b>II. Bait type, placement and non-target uptake.</b>	<b>86</b>
4.1 Introduction	88
4.2 Methods	89
4.2.1 Study area	89
4.2.2 Poison baits	90
4.2.3 Bait placement	90
4.2.4 Target versus non-target uptake	92
4.2.5 Comparison of bait type	92
4.2.6 The use of lures	93
4.3 Results	93
4.3.1 Bait placement	93
4.3.2 Target versus non-target uptake	95
4.3.3 Comparison of bait type	96
4.3.4 The use of lures	98
4.4 Discussion	98
4.5 Conclusion	100
4.6 Acknowledgements	101
4.7 References	102
<b>5. CHAPTER FIVE : Interactions between a top order predator and exotic mesopredators</b>	<b>105</b>
5.1 INTRODUCTION	107
5.2 STUDY AREA	109
5.3 METHODS	110
5.3.1 Dingoes	110
5.3.2 Cats and foxes	111
5.3.3 Data analysis	112
5.3.4 Prey abundance	113

5.4 RESULTS	113
5.4.1 Foxes	114
5.4.2 Cats	116
5.4.3 Control animals	119
5.4.4 Track transects	119
5.5 DISCUSSION	121
5.6 MANAGEMENT IMPLICATIONS	125
5.7 ACKNOWLEDGEMENTS	125
5.8 REFERENCES	126
<b>6. CHAPTER SIX: Can predator avoidance training improve reintroduction outcomes for the bilby?</b>	<b>131</b>
6.1 INTRODUCTION	133
6.2 METHODS	135
6.2.1 STAGE ONE- predator free environment	135
6.2.2 STAGE TWO- predators present	140
6.3 RESULTS	141
6.3.1 STAGE ONE- predator free environment	141
6.3.2 STAGE TWO- PREDATORS PRESENT	147
6.4 DISCUSSION	149
6.4.1 STAGE ONE- predator free environment	149
6.4.2 STAGE TWO- Predators present	151
6.5 ACKNOWLEDGEMENTS	153
6.6 REFERENCES	153
<b>7. CHAPTER SEVEN: Do release protocols influence translocation outcomes when predation risk is low?</b>	<b>156</b>
7.1 Introduction	158
7.2 Methods	160
7.2.1 Study Site	160
7.2.2 Reintroductions	161
7.2.3 Soft versus hard release burrowing bettongs	163
7.2.4 Soft versus hard release of captive and wild greater bilbies	164
7.2.5 Captive bred versus wild stick-nest rats	164
7.3 Results	166
7.3.1 Hard versus soft released burrowing bettongs	166
7.3.2 Hard versus soft released bilbies	168
7.3.3 Captive-bred versus wild stick-nest rats	170
7.4 Discussion	172
7.4.1 High predation risk	173
7.4.2 Ethical grounds	173
7.4.3 Unbounded release sites	174
7.4.4 Social or sedentary life history strategies	174
7.4.5 Supplementation	175
7.5 Conclusions	176

7.6 Acknowledgements	176
7.7 References	177
<b>8. CHAPTER EIGHT: Keep on counting: The importance of long term post release monitoring in reintroduction programs</b>	<b>181</b>
8.1 Introduction	182
8.2 Methods	184
8.2.1 Study site	184
8.2.2 Reintroductions	184
8.2.3 Population monitoring	185
8.2.4 Variables	187
8.2.5 Data analysis	188
8.3 Results	190
8.3.1 Rainfall	190
8.3.2 Track abundance	190
8.3.3 Rate of increase	193
8.3.4 Explanatory variables	196
8.4 Discussion	200
8.5 Conclusions	203
8.6 Acknowledgements	203
8.7 References	204
<b>9. CHAPTER NINE: Conclusions</b>	<b>207</b>
9.1 Guiding Principles	207
9.2 Limitations Of My Research	209
9.3 Directions For Further Research	210
9.4 Arid Zone Mammal Decline	212
<b>10. ACKNOWLEDGEMENTS</b>	<b>213</b>
<b>11. BIBLIOGRAPHY</b>	<b>214</b>

## Abstract

The Australian arid zone has the highest recent mammal extinction rate in the world with most species in the critical weight range of 35 g to 5.5 kg now regionally or globally extinct. Reversing arid zone mammal decline has become a major focus for conservation organizations and reintroduction programs are a common tool in species recovery. Unfortunately, reintroduction success in Australia is low and predation from introduced cats and foxes is commonly cited as the cause of reintroduction failure. In this thesis, I aimed to improve reintroduction success in the arid zone by exploring predation, release protocols and post release abundance at the Arid Recovery Reserve in northern South Australia. Firstly, I attempted to reintroduce threatened mammal species into both a predator free area and one where predators were controlled. Results suggested that successful reintroductions only occurred when cats and foxes were excluded. I then tested different predator reduction strategies to determine if reintroduction success could be improved, including aerial baiting, strategic bait placement and the use of a native top-order predator. Although the use of dingoes to control foxes and cats showed promise, I was unable to improve reintroduction success using poison baiting as it did not significantly reduce feral cat abundance.

I investigated the role of release strategies on reintroduction success and conducted predator avoidance training, soft and hard releases and using captive versus wild stock. Predator avoidance training did not assist long term reintroduction success of the bilby but some behavioural differences were detected. Results suggest that Australian arid zone species may be able to learn predator avoidance behavior but this may not necessarily translate into improved reintroduction outcomes. The use of soft and hard releases and captive and wild stock had little effect on reintroduction success when cats and foxes were excluded. Interspecific differences in post-release mortality and behaviour indicated that soft releases may be useful at unrestricted release sites, in situations of high predation risk and where social, sedentary species which invest heavily in their shelters are being released.

Finally, I analysed long term monitoring data for four reintroduced threatened species to determine whether factors such as rainfall, time since release or temperature influenced post-release population fluctuations. Factors significantly influencing abundance included the Indian Ocean Dipole and temperature. Time since release was still the most important factor influencing abundance even 10 years after release indicating that reintroduced populations may not stabilize for decades and long term monitoring is essential.

Regardless of reintroduction protocols, new methods of broadscale cat control are required before broadscale reintroduction success can be improved in the Australian arid zone. Present control methods are insufficient to enable successful reintroductions of cat-sensitive mammal species without exclusion fencing. However, exclosures are relatively small and expensive, and can create problems such as overstocking. Future arid zone reintroductions should focus on broadscale reintroductions without fences to ensure widespread recovery but this will require the development of improved cat control methods. Species-specific predator thresholds are also needed to trigger management actions and improve the predictability of reintroduction outcomes.

# 1. CHAPTER ONE : LITERATURE REVIEW AND CONTEXTUAL STATEMENT

## 1.1 MAMMALIAN EXTINCTIONS IN ARID AUSTRALIA

Australia has the highest contemporary mammal extinction rate in the world with 18 species becoming extinct over the last 200 years (Short and Smith 1994). The range of a further 26 species has declined significantly and many species are now restricted to off-shore islands. The arid interior of Australia has been the worst affected region, where most medium-sized mammals are now locally or globally extinct (Burbidge and McKenzie 1989). These mammals include a variety of wallabies, bandicoots, rodents and dasyurids, once common across the arid stony and sand plains of inland Australia. South Australia has the highest mammalian extinction rate of any state (Kemper 1990), 22% of mammal species have disappeared since European settlement (Kemper 1990). Reptiles and birds have suffered much less than their mammalian counterparts (Morton 1990) and no arid zone reptile species has been known to become extinct since European settlement.

There have been three waves of mammalian extinctions in Australia with the first two occurring in the Pleistocene and Holocene, and a more recent catastrophic decline around 100 to 200 years ago coinciding with the arrival of Europeans in Australia (Johnson 2006). During the post-European extinctions, many species became extremely rare or functionally extinct by the 1930's but some species such as the burrowing bettong (*Bettongia lesueur*) persisted until the 1950's or even 1960's. Burbidge and McKenzie (1989) introduced the concept of 'Critical Weight Range' mammals after noting that most extinctions occurred in non-flying mammals weighing between 35g and 5.5kg. Johnson (2006) summarised the post-European extinctions as affecting mainly medium-sized mammals in the southern arid zone, occurring over a long time period from south to north and east to west, and affecting smaller species before larger species.

Many reasons have been suggested for the cause of mammalian extinctions in arid Australia including predation from introduced predators, competition with introduced herbivores, hunting, poisoning, exotic diseases, habitat clearance and changes to fire regimes (Jones 1924, Finlayson 1961, Newsome 1971, Burbidge and Fuller 1979, Burbidge and McKenzie 1989, Friend 1990, Kemper 1990, Short 1998). Causes of mammalian extinctions have been the subject of lively debate with the most popular theories citing a combination of causes.

Morton (1990) introduced the refugia concept which suggested that herbivorous and omnivorous mammals were disproportionately impacted by droughts in largely infertile deserts and that during droughts these species were restricted to scattered more fertile areas. Contracting back to small pockets increased their vulnerability to extinction and when introduced herbivores such as rabbits and stock arrived in Australia the composition of the vegetation was significantly altered in the refuges that these animals used during drought. This, coupled with introduced predators and altered fire patterns, resulted in extinctions during major drought events.

Other authors support the theory that habitat degradation from herbivores such as the European rabbit (*Oryctolagus cuniculus*) and domestic stock were the main cause of decline. Jones (1924) and Newsome (1971) cite competition from rabbits for food and burrows as a reason for loss of native species. However Robley *et al.* (2002) found no influence of rabbit



density on the survival, recruitment or rate of increase of reintroduced burrowing bettongs. Seebeck (1979) suggests that physical change in soil structure due to trampling of stock may have contributed to the decline of the eastern barred bandicoot (*Perameles gunnii*) and Lunney (2001) also concluded that overstocking caused the rapid demise of small-medium sized mammals in western NSW. Copley (1999) supports Morton's (1990) refugia hypothesis and suggests that the local extinction of stick-nest rats (*Leporillus spp*) in the Murray-Darling confluence and the Flinders Ranges occurred by about the 1860's and 1870's, well before the arrival of rabbits and foxes and attributes the decline to overgrazing by sheep. He cites habitat degradation by sheep and rabbits as the primary cause of the decline in the greater stick-nest rat during major drought events. Predation by both native and introduced predators rather than introduced predators *per se* is thought to be a secondary cause of extinctions due to vulnerability during drought.

However, in central Australia where populations of many species persisted until the 1930' and 1940's and in many cases where pastoralism had never occurred, population crashes of mammals occurred soon after the arrival of the red fox (*Vulpes vulpes*) (Finlayson 1961; Copley 1999) but 30 years after the arrival of rabbits. Short (1998) also concluded that foxes caused the extinction of rat-kangaroos in New South Wales by examining bounty payments. Similarly, foxes first appeared near Hamilton between 1906 and 1914 and this event was associated with the deaths of many eastern barred bandicoots (Seebeck 1979).

Some studies have attempted to model species decline against a range of parameters. One such study by Smith and Quin (1996) suggested that fox and rabbit presence was a major predictor of decline in Australian rodents with cats (*Felis catus*) the best predictor of decline in small <35g conilurine rodents. Their "hyperpredation" model suggests that introduced predator levels are elevated and maintained at high levels where rabbits and introduced mice are present causing declines and extinctions in 'vulnerable' local native rodent species. Vulnerable species include those with low reproductive rates, species within the preferred prey size ranges of predators and those species that live in areas that lack refuges such as burrows, rockpiles and trees. Other species may co-exist with predators and rabbits if they have high reproductive rates similar to rabbits, and can bounce back quickly when predation eases. The emphasis of this hypothesis is on predation by introduced predators as the major cause of decline, a concept supported by Johnson (2006) who reviewed the history of decline in Australian mammals. Courchamp *et al.* (2000) surmised that hyperpredation is likely to cause the extinction of a native mammal prey species if the introduced prey species has a higher population density, higher population growth rate or is harder to catch than the native prey species. The introduced European rabbit has a higher reproductive rate than native Australian mammals and is capable of rapid population growth, traits which suggest it may be an agent for hyperpredation in the Australian arid zone.

Prey-switching during drought or times of food shortage is another popular concept in mammalian extinction theory and a component of the hyperpredation model. This theory suggests that introduced herbivores such as rabbits led to maintenance of artificially high cat and fox densities that switched to native mammalian prey during droughts when rabbit numbers declined. Two major droughts have been documented during the 20<sup>th</sup> century in the arid zone, 1925-38 and 1958-64 (Griffen and Friedel 1985). The drought in the 1930's coincides with the dramatic decline and extinction of many arid zone species. Wet conditions in the early 1920's prior to the drought led to a build up in rabbit numbers that crashed during the subsequent drought (Copley 1999). Prey switching by foxes and cats after the rabbit crash may have contributed to the extinction of small mammals at that time.

Further evidence to support the hyperpredation and prey switching models comes from Macquarie Island, where Taylor (1979) found cat predation caused extinction of the Macquarie Island Parakeet (*Cyanoramphus erythrotis*) following the introduction of rabbits. Cats were present on the island prior to the introduction of rabbits and co-existed with the parakeets for more than 60 years. Co-existence was thought to be due to a density –dependent response to a shortage of prey during winter. However, when rabbits were introduced they provided a year round food supply for the cats leading to an increase in cat abundance and abnormally high predation on secondary indigenous prey. Within 20 years the parakeets were extinct. Prey switching is also thought to have led to the extinction of the Davenport Range black-flanked rock wallaby colony (*Petrogale lateralis*) after the introduction of calicivirus in 1996 (Moseby *et al.* 1998).

Areas where threatened species populations remain extant on the mainland also appear to be influenced by the absence of the fox and feral cat. In areas such as the north-west Kimberley and the northern Tanami Desert, populations of some threatened species and medium-sized native mammals are still relatively intact, possibly due to the fact that the red fox and European rabbit are not established there (King and Smith 1985). Remnant populations of the brush-tailed bettong (*Bettongia penicillata*) in Western Australia increased significantly after the initiation of fox baiting (Kinnear *et al.* 2002). The introduction of cats to Hermite Island led to extinction of both the golden bandicoot (*Isoodon auratus*) and spectacled hare wallaby (*Lagorchestes conspicillatus*) (Burbidge 1971).

An insight into the role of cats and foxes in the demise of native mammals can be gained from attempts to reintroduce them into the wild. Reintroductions are a relatively recent tool used to halt or reverse the decline of threatened species in Australia. Reintroductions can also act as an insurance policy, reducing the risk of a single catastrophic event causing the extinction of a species with a limited distribution. Many attempts have been made to reintroduce medium-sized mammals including greater stick-nest rats, burrowing bettongs, brush-tailed bettongs, golden bandicoots and greater bilbies (*Macrotis lagotis*) into arid and semi-arid Australia. Most reintroduction attempts have failed, primarily due to predation from introduced predators such as cats and foxes (Short *et al.* 1992; Christensen and Burrows 1994; Gibson *et al.* 1994; Southgate 1994; Southgate and Possingham 1995; Priddel and Wheeler 2002). The most successful reintroductions have been onto islands or predator-free enclosures on the mainland (Richards and Short 2003; Arid Recovery 2008). Richards and Short (2003) reported successful reintroductions onto the mainland at Heirisson Prong in Western Australia when cats and foxes were excluded but rabbits were still present. However, fox incursions were considered to be responsible for significant mortality of reintroduced bettongs after release (Short and Turner 2000).

A relatively new theory of mammalian extinction highlights the role of the dingo (*Canis lupus dingo*) in protecting threatened species. Smith and Quin (1996) found lower rates of conilurine rodent extinction in areas where dingoes were abundant, and Johnson (2006) has suggested that mammal extinctions and decline are less severe in areas where dingoes are still present and that dingoes may have a positive influence on threatened species by suppressing cat and fox populations. This increasingly popular opinion suggests that dingo populations have a net benefit to wildlife because they have a negative impact on the red fox (Smith & Quin 1996; Hobbs 2001; Newsome 2001; Newsome *et al.* 2001; Daniels & Corbett 2003). The mesopredator release hypothesis (MRH) predicts that reduced abundance of top-order predators results in increased abundance or activity of smaller predators such as feral cats and foxes and consequently has detrimental impacts on the prey of the smaller predators (Crooks & Soule´ 1999). In North America, where coyote (*Canis latrans*) abundance has declined, red fox numbers have increased (Goodrich & Buskirk 1995; Crooks & Soulé 1999). In Australia,

Letnic (2007), Letnic *et al.* (2009) and Wallach *et al.* (2010) also favour the MRH as well as the trophic cascade theory which suggests that top predators such as dingoes have both positive and negative effects on lower trophic levels and may indirectly enhance plant biomass (Hairston *et al.* 1960). The removal of dingoes may thus allow exotic herbivores and smaller introduced predators to increase, depleting herbivorous food supplies, increasing predation pressure and increasing risks of transferring exotic diseases such as toxoplasmosis and mange to native species.

Unfortunately there is little experimental evidence to support the perceived role of the dingo in suppressing fox and cat abundance at landscape scales (Mitchell and Banks 2005) with only circumstantial or anecdotal evidence that relies on historical or observational data. However dingoes have been recorded directly preying on cats (Palmer 1996a and 1996b, Paltridge 2002) as well as excluding them from resource points such as carcasses during drought (Pettigrew 1993; Corbett 1995).

## 1.2 MAMMALIAN REINTRODUCTIONS

Reintroductions are commonly used in an attempt to re-establish threatened species in areas where they were once present but have declined or become locally extinct. Reintroductions are most commonly used for mammalian species in arid Australia, although some bird and reptile reintroductions have been attempted (Sedden *et al.* 2005; Read *et al.* 2011). Reintroductions assume the previous presence of the species in an area, which is usually verified through subfossil records, museum records, historical records or local knowledge.

Successful reintroductions require that the threatening processes leading to the original decline have been removed or controlled, something that is often difficult to confirm if the precise reason(s) for the decline is (are) unknown. Reintroductions often also require additional pre and post release habitat modification and management as in many cases habitats have been altered significantly since the arrival of Europeans.

There are three common approaches to reintroductions in Australia. The enclosure approach involves reintroducing animals to a fenced pen where feral animals, in particular predators, have been removed. Examples of this approach include Warrawong Sanctuary, the Arid Recovery Reserve, Yookamurra Sanctuary and Venus Bay Conservation Park in South Australia, Heirisson Prong in Western Australia, Scotia Sanctuary in NSW and Currawinya Reserve in Queensland. Enclosures are of varying sizes and some take advantage of peninsulas to minimise fencing requirements (Coman and McCutchen 1994; Long and Robley 2004). Some enclosures still contain populations of feral species such as rabbits.

The second approach involves using islands where threatening processes such as predators or competitors have been removed or are absent (Burbidge 1989; Abbott 2000). Cats have been successfully eradicated from several Australian islands (Copley 1991; Burbidge 1989; Twyford *et al.* 2000) as well as islands in New Zealand (Veitch 1985) and the sub-Antarctic (Bester 1993). Islands such as Reevesby Island and Thistle Island in South Australia and Faure Island and Salutation Island in Western Australia have been used to house populations of threatened species. Poison baiting for feral cats has been most successful in confined areas such as islands (Twyford *et al.* 2000) or in areas where alternative prey such as rabbits are in low abundance (Algar *et al.* 2007). The use of islands minimises the risk of incursions from feral species but islands of a suitable size and habitat type are limited, often heavily altered by stock grazing and not always available for use.

The third approach is to attempt to control predators and other threats on the mainland without the use of fences. These projects generally use broadscale poison baiting to reduce fox and/or cat numbers and usually have much lower success rates than reintroductions to islands and exclosures. Bounceback in South Australia and Western Shield in Western Australia are two examples of reintroduction attempts using broadscale predator control on the mainland. The advantage of this method is that large areas can be potentially managed for threatened species at a comparatively lower cost. Although there is evidence to suggest this method may be successful (Kinnear *et al.* 2002) the long term benefits to threatened species are yet to be proven.

### 1.2.1 CRITERIA FOR SUCCESS OF MAMMALIAN REINTRODUCTIONS

A successful reintroduction is usually considered to be one where a genetically viable population of the re-introduced species is established and persists in the long term. Genetic viability is subjective and whilst some practitioners advocate a minimum population size of 500 to maintain additive genetic variance (Falconer 1960; Franklin 1980), Lande and Barrowclough (1987) revised this to a few hundred animals. However these researchers did not include actual numbers of alleles, a factor that Denniston (1978) and Gale and Lawrence (1984) consider could be important for long term conservation. To maintain sufficient numbers of alleles in a population would require much larger population sizes.

The effective size of a population is different to the actual size and includes the number of individuals that contribute equally to the next generation of parents. The effective size is only the same as the actual number of animals present if a number of conditions are met such as high dispersal, random mating, no fluctuations in census size, equal sex ratio and no overlap in generations (Sherwin and Brown 1990). Sherwin and Brown (1990) estimated the eastern barred bandicoot census size to be 633 but with an effective population size of only 67, considered ineffective for long term conservation by Franklin (1980) and Lande and Barrowclough (1987). However, some species can retain a normal level of variation with very low effective sizes (James 1982 in Sherwin and Brown 1990). Sherwin and Brown (1990) suggested that controlling predators that prey on young animals may reduce the variation in pouch young survival and effectively increase the effective population size relative to census population size.

Other conservationists use tools such as Population Viability Analysis (Shaffer 1990) to determine the probability of a population becoming extinct. This system models and analyses the various deterministic and stochastic forces determining the fate of small populations. Stochastic components can include demographic stochasticity which is the fluctuation in population size resulting from individual reproductive or mortality events (finding a mate or being killed by a predator for example), environmental stochasticity (caused by random fluctuations in the environment – rainfall, temperature etc), catastrophic stochasticity (e.g. hurricanes, prolonged drought, large wildfires), genetic stochasticity (random genetic change such as increased homozygosity leading to reduced fecundity and lower genetic variation).

Many practitioners set *a priori* criteria for reintroduction success (Backhouse *et al.* 1994, Short and Turner 2000; Richards and Short 2003; Vale *et al.* 2004) such as a percentage survival after 12 months, reproduction within 6 months, wild-bred animals reproducing within a certain number of years, population persistence after 5 years and total number of animals alive within one to five years of reintroduction. Often these criteria are subjective and have little bearing on the long term persistence or extinction of the population but they allow short-

term success to be measured and ensure monitoring is conducted to enable success criteria to be assessed. Short and Turner (2000) chose the criteria “more than 265 bettongs within 5 years” as a measure of success based on the estimated population at which a maximum sustained yield can be harvested from the population by a predator. The high variability of arid environments suggests that additional criteria incorporating survivorship or persistence after drought should be incorporated into success criteria. The persistence of source populations after removal of animals for release should also be considered an essential criterion of a successful reintroduction.

There are many factors that can contribute to reintroduction success or failure. These can be divided into intrinsic and extrinsic effects and both can have a significant influence on reintroduction outcomes. Intrinsic factors are those that can largely be controlled or managed by the reintroduction practitioner including release protocols such as hard versus soft releases, pre-release training such as predator avoidance, the number of animals released, genetics and composition of source population, artificial placement of shelter sites and release timing and order. Extrinsic factors include those that are stochastic or external and are difficult to control, including the influence of predation and/or competition with *in situ* species and the effect of temperature, rainfall and time since release. Intrinsic and extrinsic factors should not necessarily be viewed as independent, for example the success of predator-training, an intrinsic factor, can be heavily dependent on the extrinsic predation level that is present at the time of release.

### *1.2.2 INTRINSIC FACTORS AFFECTING REINTRODUCTION SUCCESS*

#### **1.2.2.1 Hard versus soft release**

Reintroductions can be categorised as ‘hard’ or ‘soft’ releases depending on whether assistance is provided to the animal at the time of release. A hard release involves releasing an animal directly into the wild without any form of external assistance from the practitioner. A soft release involves providing assistance in the form of food, shelter, acclimatisation pens etc. Many researchers believe reintroduction success can be improved using soft release techniques.

Short and Turner (2000) and Richards and Short (2003) considered “the refuge”, a small 8 ha pen that was used as an initial release site, to be a key factor in the success of burrowing bettong and western barred bandicoot reintroductions at Heirisson Prong. The reasons cited were for initial containment and protection from fox incursions in the early stages of reintroduction. A release pen may also allow competitors such as rabbits and native predators such as goannas (*Varanus* spp.) to be removed (Richards and Short 2003), possibly improving initial release success.

Some reintroduction studies have encountered problems with dispersal after release, particularly of males, and release pens may assist with population establishment by preventing large scale dispersal. During a golden bandicoot reintroduction to the Gibson Desert, 85% of bandicoots moved up to 4 km from the release site within one week of release (Christensen and Burrows 1994). Several male burrowing bettongs released into Heirisson Prong in W.A. were recorded moving more than 10km from the release site (Short and Turner 2000) with one individual moving 21km. Males that disperse large distances from the release site are unlikely to contribute to the population. Additionally, researchers have found that in general there is a positive linear relationship between mobility and predation risk (Norrdahl and Korpimäki 1998). Some studies have recorded reduced dispersal when males are released

into already established populations where females are present (Soderquist 1994) or when males and females are released at the same time and there was an established population already at the release site (burrowing bettongs -Short and Turner 2000, western barred bandicoots- Richards and Short 2003). Short and Turner (2000) found no difference in burrowing bettong survival when comparing animals released into familiar (adjacent to “refuge”) and unfamiliar environments (2.5km away).

Other advantages of soft release pens include the ability to return animals to the pen if they lose weight after release and allowing natural dispersal if the pen mesh size is large enough to allow juveniles to escape. Richards and Short (2003) found juvenile western barred bandicoots could disperse from the refuge through the wire mesh, and adults also escaped. Some animals are better suited to soft release than others as soft-release pens can also have negative influences due to overcrowding and intraspecific aggression. Scarring and tail loss were recorded in western barred bandicoots even when only 10 animals were contained within a 17ha pen (Richards and Short 2003). Other researchers have also experienced problems with bandicoots fighting when too many individuals are housed together (Lyne 1982).

Providing supplementary food and water after release may assist with preventing initial post-release weight loss and is a common method used by many reintroduction practitioners (e.g. Southgate *et al.* 1994). The amount of time that food and/or water is provided varies considerably with some studies finding no change in weight or condition when it is finally removed (western barred bandicoots- Richards and Short 2003).

### **1.2.2.2 Pre-release training**

Some captive-bred animals or animals isolated from predators either evolutionarily or throughout their lifetime (ontogenetic) may no longer express appropriate antipredator behaviour and are therefore unable to survive their first predator encounter (Griffin *et al.* 2000). This is a problem for many Australian native mammals as few mammalian predators existed naturally in Australia before the introduction of the feral cat and red fox. Unless the reintroduced animals have survival skills not available to the original population, or the new predator(s) have been controlled or eliminated, reintroduction programs generally fail (McLean *et al.* 1996).

Consequently, increasing interest in predator avoidance training has seen experimental trials involving a range of taxa including fish (Magurran, 1989), birds (McLean *et al.* 1999) and mammals (Mineka and Cooke, 1988; Griffin *et al.* 2000; McLean *et al.* 2000), with recent studies focusing on Australian species. Empirical evidence from such studies suggests that training can improve antipredator skills. Griffin *et al.* (2000) found that the tammar wallaby (*Macropus eugenii*) could be trained to respond to a model fox by associating it with a simulated capture. Interestingly, tammar wallabies responded similarly to a model cat despite the fact that no aversion event was paired with the cat’s presence. Rufous hare-wallabies (*Lagorchestes hirsutus*) were conditioned to fear a model fox after its presence was paired with a loud noise and wallaby alarm calls or squirts from a water pistol (McLean *et al.* 1996). Captive raised Siberian polecats (*Mustela eversmanni*) showed increased alert behaviour after the presence of a model owl and badger were paired with an aversion event (animals shot with elastic bands), with individuals reacting fearfully after just one training event (Miller *et al.* 1990). Some attempt was made to expose burrowing bettongs to a predator to reduce their naivety at Heirisson Prong in the early years of the reintroduction but this was later abandoned when more effective predator control was implemented (Short and Turner 2000).

Several researchers have also tried unsuccessfully to use non-predatory stimuli such as goats (*Capra aegagrus hircus*), conspecifics (Griffin *et al.* 2002) and even a bunch of flowers (Mineka and Cook 1988) to elicit a fear response. This suggests that many species are able to discriminate between predators and non-predators and that some species may have a predisposed ability to acquire fear of predators (Griffin *et al.* 2002). Research into the use of olfactory cues in predator avoidance training has shown that in most studies, predator-experienced species changed their behaviour when faced with the scent of known predators but predator naïve species did not (review by Blumstein *et al.* 2002). Blumstein *et al.* (2002) suggested that species isolated from predators over an evolutionary time period or during an animal's lifetime may lose the capacity to recognise the olfactory cues of predators and may have to learn to recognize such cues.

Although several studies have demonstrated that predator-avoidance training can lead to a change in behaviour of the prey animal, few if any studies have compared the survival of trained and untrained animals after reintroduction into the wild. Additionally, it is not known if any trained behaviour of adult animals is successfully transferred to offspring after reintroduction.

Another form of pre-release training that may improve reintroduction success was suggested by Banks *et al.* (2002) who found that animals with low mobility after release were associated with higher concentrations of odour waste. Odour wastes are attractive to predators and higher predation rates were recorded in *Microtus* voles when individuals remained close to their release site immediately after release. Banks *et al.* (2002) suggests pre-release training of animals may be needed to overcome initial post-release site fidelity caused by fear of new environments and encourage movement away from the release site.

### **1.2.2.3 Release size and composition**

The size and composition of release groups varies considerably between reintroduction programs but the majority of bird and mammal reintroductions in Australia and overseas between 1973 and 1986 have comprised less than 75 animals (Griffith *et al.* 1989). Small founder groups can increase the risk of extinction from both genetic inbreeding and stochastic factors. Sinclair *et al.* (1998) modelled predator-prey interactions and found there are thresholds of population density that enable reintroduced populations to cope with exotic predators. However most founder sizes used in reintroductions are significantly below this size and release sizes of hundreds of animals may be required in areas where exotic predators are present (Sinclair *et al.* 1998). This point is highlighted by the founder sizes of reintroduced eastern barred bandicoot populations in Victoria. Founder sizes comprise 50-130 individuals but few if any populations are secure due to drought and predation by introduced predators (Watson and Halley 2000). Matson *et al.* (2004) found large release size to be a significant indicator of successful reintroductions of the black-faced impala (*Aepyceros melampus petersi*). Conversely, successful reintroductions have been recorded using small founder groups in predator-free environments e.g. a reintroduction of nine western barred bandicoots at Heirisson Prong increased to 130 in just four years (Richards and Short 2003).

Founder size is important but habitat quality and control of threatening processes may be more important (Griffith *et al.* 1989; Caughley and Gunn 1996). Modelling by McCallum *et al.* (1995) found that in almost all circumstances, a single reintroduction of bridled nailtail wallabies (*Onychogalea fraenata*) is preferable to multiple releases of the same number of animals. This is due to the fact that predator-prey theory predicts that predation will have a much larger effect on small populations.

### 1.2.3 EXTRINSIC FACTORS AFFECTING REINTRODUCTION SUCCESS

#### **1.2.3.1 Predation**

Predation from both native and exotic predators can have a major impact on the success of reintroductions. Although strong evidence for the role of cats in the extinction of arid zone mammals is scarce, feral cats are thought to cause the decline and extinction of many native animals on islands (Dickman 1996). Priddel and Wheeler (2002) found cat predation responsible for the failure of brush-tailed bettongs to re-establish at Yathong Nature Reserve in western NSW. Cats arrived in Australia during European settlement (Abbott 2002) and were widespread across the continent soon after the 1880's. They were known to be sympatric with some species such as western barred bandicoots on the Nullarbor in Western Australia in the early 1900's (Richards and Short 2003). The eastern barred bandicoot is still present in Tasmania where cats and rabbits are present but is known to be preyed on by both domestic and feral cats (Booth and McCracken 1994). The re-introduced western barred bandicoot population at Heirisson Prong managed to increase and colonise the 12 square km release area even with 2-3 cats (1 per 4-6 square km) present on the peninsula over 2 years. However densities of western barred bandicoots were estimated at less than a quarter of that recorded on the source islands of Bernier and Dorre (Richards and Short 2003) and cats may have contributed to the high mortality rate of bandicoots recorded (Richards and Short 2003). Short and Turner (2000) suggest that feral cats, although preying on some adult reintroduced burrowing bettongs, are more likely to affect recruitment by preying on juveniles. Recruitment at Heirisson Prong grew sharply when feral cats were removed.

The red fox arrived much later in the arid zone than the feral cat, with foxes first released in Victoria in the 1860's (Catling and Coman 2008) and reports of foxes in northern South Australia occurring from about 1905. Western barred bandicoots and burrowing bettongs disappeared on the mainland within 10-20 years of fox establishment and 30-40 years after rabbits (Richards and Short 2003; Short and Turner 2000). The presence of foxes appears to have a major impact on remnant and reintroduced populations of some mammal species, including eastern barred bandicoots in Victoria (Short *et al.* 2002) and black-flanked rock wallabies (*Petrogale lateralis*) in Western Australia (Kinnear *et al.* 1988). Short *et al.* (2002) found reintroduced burrowing bettong mortality rates of 77%, 36% and 46% with each of three fox incursions into the Heirisson Prong enclosure in Western Australia.

#### **1.2.3.2 Competition**

Competition from introduced herbivores such as rabbits is thought to be at least partly responsible for the decline and extinction of arid zone mammals (Morton 1990). Rabbits alter habitat by selective browsing and also elevate feral predator abundance. Rabbits provide food and shelter (burrows) for feral cats (Taylor 1979; Newsome 1990) and young rabbits are particularly prevalent in their diet (Catling 1988, Read and Bowen 2001). Reintroductions into areas where rabbits are still present have been successful (Richards and Short 2003) but have often led to plague increases in rabbit numbers due to the removal of exotic predators. Newsome *et al.* (1989) also found rabbit abundance significantly increased in arid areas when cats and foxes were controlled. Richards and Short (2003) found litter size in reintroduced western barred bandicoots increased with a decrease in rabbit abundance suggesting rabbits may have some effect on reproductive output. However, Short and Turner (2000) suggested that the threat of rabbits to the reintroduction of burrowing bettongs was substantially less



than that posed by feral cats and foxes and Robley *et al.* (2002) found no influence of rabbits on the survival, recruitment or rate of increase of reintroduced burrowing bettongs. The main threat to reintroduced species from rabbits appears to be the secondary influence of sustaining higher predator numbers which in turn prey on native species. However, removal of feral cats and foxes but not rabbits could also negatively affect reintroduction success as increased rabbit abundance could possibly lead to habitat damage and starvation for reintroduced species.

### **1.2.3.3 Temperature and rainfall**

Local weather and seasonal conditions before, during and after release are likely to influence the survival of reintroduced populations. Rainfall is the main driver of arid zone systems and temperature extremes are also common in desert environments. By reintroducing species just prior to or during their breeding season, maximum population increase may be achieved during the early stages of reintroduction when the chances of mortality are high. Although breeding in mesic species such as eastern barred, southern brown (*Isoodon obesulus*) and northern brown (*I. macrourus*) bandicoots is often correlated with rate of change of minimum temperature and daylength (Barnes and Gemmell 1984), rainfall is the main stimulus of reproductive activity in many arid zone species. Significant rainfall events lead to an abundance of food resources for arid zone mammal species, many of which can breed continuously when conditions are favourable (Tyndale-Biscoe 1968). Some arid zone mammal species also have flexible breeding seasons, Richards and Short (2003) found the peak breeding season in western barred bandicoots is June to September on Bernier and Dorre Islands but breeding may extend into summer when there is above average spring and summer rainfall.

In arid zone areas where rainfall is unpredictable and aseasonal it may be difficult to time reintroductions to coincide with rainfall events and maximise population increase. However, it is possible to avoid releasing animals during drought conditions which can cause serious significant population declines or even localised extinction. Western barred bandicoots declined significantly on Dorre Island during a prolonged drought from 1986 to 1989 and eastern barred bandicoots near Hamilton declined during a drought in 1966-68 and were only found in areas of permanent springs and streams (Seebeck 1990).

High summer temperatures in arid areas could also influence post-release population dynamics, particularly in mammal species that live above ground or rely on building nests or burrows for shelter. Releasing animals during or just prior to the summer months may mean individuals have not had time to construct suitable nests or burrows to protect them from high temperatures. Even for established populations, high summer temperatures could cause population declines through lower reproductive output, deaths from heat exhaustion or increased predation from native reptiles such as goannas and snakes.

### **1.2.3.4 Time since release**

Some species undergo a characteristic pattern of fluctuating abundance after release which includes a latent establishment phase, an exponential increase phase, a significant decline or 'crash' phase followed by a more consistent and lower population level. Stick-nest rats reintroduced to Reevesby Island increased significantly for the first five years after release before undergoing a significant population crash. The population crash occurred after severe vegetation damage from overbrowsing and is likely to have been caused by nutritional stress. The population is now considered to be at carrying capacity and the vegetation has recovered

to some extent (J. Van Weenen pers. comm.). Brush-tailed bettongs also exhibited exponential growth after release to Wedge Island before a population crash reduced the numbers to much lower levels (J. Van Weenen pers. comm.). However, this pattern is not evident in all reintroduced species or all reintroductions of a particular species and may be dependent on location, the presence or absence of predators, ability of the animal to disperse, food supply and/or size of the release area.

#### 1.2.4 POST-RELEASE MONITORING

Although monitoring does not have a direct bearing on reintroduction success it is an important tool for assessing and evaluating reintroduction outcomes. Post-release monitoring is an important component of any reintroduction program but is often the most neglected. Long term monitoring may increase our knowledge of the ecology of threatened species and assist in formulating future release protocols. Monitoring methods are not standardised in Australia and researchers use a variety of methods including radiotracking (Short and Turner 2000), track monitoring, remote cameras, trapping (Short and Turner 2000), scanning plates, and spotlighting. Capture-mark-recapture is often used to determine population size (e.g. Short and Turner 2000) but distance sampling is becoming more popular (e.g. Scotia Sanctuary, Australian Wildlife Conservancy).

Radio-collaring has been used extensively on eastern barred bandicoots but many problems have been reported including feet caught in collars and neck ulceration (White and Garrott 1990; Booth and McCracken 1994; Seebeck and Booth 1996). Richards and Short (2003) reported collaring problems during the initial release of western barred bandicoots but not subsequently, possibly due to only collaring for less than 14 days and continual monitoring during that period for weight loss and chafing. Long-nosed bandicoots (*Perameles nasuta*) have been successfully radio-collared for periods of up to 14 days (Chambers and Dickman 2002) and 6-8 weeks (Scott *et al.* 1999).

### 1.3 CONTEXTUAL STATEMENT

The Australian arid zone has experienced the highest recent extinction rate in the world and the worst success rate for reintroductions. It is clear from previous arid zone reintroduction studies that predation from introduced foxes and cats are the major determinants of reintroduction success or failure in arid Australia. The development of new techniques for predator control and/or the improved predator awareness of naive native species are likely to be critical elements required to improve reintroduction success. Other factors that appear to be important include reintroduction protocols such as soft releases and release size. Little is known about the influence of factors such as temperature and rainfall, despite the fact that in arid environments these are major productivity drivers. The objective of this thesis is to explore intrinsic and extrinsic factors that influence the success of mammalian reintroductions in arid South Australia and ultimately improve reintroduction outcomes. This thesis is divided into four sections: The first three sections on predation, reintroduction protocols and post-release population dynamics are considered key determinants of reintroduction success. The final section is the conclusion chapter.

The first section of this thesis compares the success of mammalian reintroductions using two already established methods of feral animal control; exclusion fencing and integrated predator control (poisoning, shooting, trapping). The success or failure of these reintroductions is discussed in relation to predator abundance counts. Additionally, two relatively new methods of broadscale feral animal control; aerial baiting and the use of dingoes, are also investigated and their merits discussed.

Secondly, reintroduction protocols to maximise post release survival are investigated, including hard and soft releases, captive versus wild source populations and predator-awareness training. The fates of reintroduced animals exposed to these different treatments are compared. Results are used to suggest optimum reintroduction protocols for arid zone threatened species.

Thirdly, the post-release population dynamics of four re-introduced species and the influence of extrinsic factors such as season, rainfall and time since release are investigated. Results from four species reintroduced over 10 years are compared with other arid and mesic release sites and used to predict the success of future reintroduction programs in arid Australia. Where possible, management actions required to ensure the persistence of self-sustaining populations are also suggested. This is particularly important in arid areas where temperature and rainfall extremes exist and where few reintroductions have been successful.

Finally, the results from the previous three sections are synthesised in a conclusion chapter which also include directions for future research.

This thesis consists of seven research chapters and a conclusions chapter. Five of the research chapters (chapters 2-6 inclusive) had been published in Australian and international journals at the time of thesis submission with a further chapter (chapter 7) currently under review. References are presented at the end of each research chapter and all references used in the thesis are included in a bibliography at the end of the thesis. Table and figure numbers and reference formats are independently assigned to each chapter and are based on individual journal requirements. The overall theme of the thesis and chapter summaries are outlined below.

**Objective: To improve mammalian reintroduction success in the Australian arid zone.**

**Aims: To explore intrinsic and extrinsic factors related to reintroduction success and assist in the development of:**

- 1) effective methods of broadscale predator control;
- 2) reintroduction protocols and post-release monitoring;
- 3) predictive tools for post-release population dynamics;

### *1.3.1 THE INFLUENCE OF PREDATION*

#### **Chapter 2- Predation determines the outcome of 10 reintroduction attempts in Arid South Australia**

This section compared the success of two common methods of reintroduction in Australia; exclosures and broadscale control. Reintroduction success was compared within the fenced Arid Recovery Reserve and the Wild West Zone, an area of unfenced arid zone habitat adjacent to the Arid Recovery Reserve. Predators were excluded within the Arid Recovery Reserve by a 1.8m high fence whilst intensive predator control in the Wild West Zone was achieved through poison baiting, trapping and shooting. Ten reintroduction attempts over ten years were attempted. This chapter has been published in the journal, *Biological Conservation*.

Moseby, K.E., Read, J.L., Paton, D.C., Copley, P., Hill, B.M. and Crisp, H.M. (2011). Predation determines the outcome of 11 reintroduction attempts in arid Australia. *Biological Conservation* **144**, 2863-2872.

#### **Chapter 3- The use of poison baits to control feral cats and red foxes in arid South Australia. I. Aerial baiting trials.**

The red fox has been successfully controlled in many areas of Australia using poison meat baits (Thomson and Algar 2000) but poisoning feral cats has often been less effective owing to poor bait uptake (Risbey *et al.* 1997; Kinnear *et al.* 1998; Burrows *et al.* 2003; Algar and Burrows 2004; Hegglin *et al.* 2004; Olsson *et al.* 2005; Algar *et al.* 2007; Moseby *et al.* 2009). Poison baiting has a long history in Australia, with most practitioners now using the poison 1080 (sodium monofluoroacetate), a derivative of the naturally occurring fluoroacetate compound found in many *Gastrolobium* and *Oxylobium* plants in Australia (Eason 2002). The 1080 compound is odourless, tasteless and colourless and many native species have a high tolerance to it. The poison is injected into a bait substrate which is normally meat-based. A new predator poison has recently been developed (PAPP) which is considered more humane than 1080 but it is still in the experimental trial stage and not available for broadscale use.

This chapter outlined an attempt to control feral cats and foxes at a landscape scale using the recently-developed cat bait, 'Eradicat', from Western Australia (Algar *et al.* 2007). Although the 1080 Eradikat bait was developed to target feral cats, it is also highly effective against foxes (Algar and Burrows 2004). The baits were dispersed aerially using a helicopter or plane and bait density, timing and bait area were manipulated to determine baiting success. This chapter has been published in the journal *Wildlife Research*.

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

#### **Chapter 4- The use of poison baits to control feral cats and red foxes in arid South Australia II. Bait placement, lures and non-target uptake.**

This chapter followed on from chapter three and investigated whether bait uptake could be improved using lures or by varying bait placement. The non-target uptake of poison baits was also studied. Of particular importance was whether threatened or reintroduced species were likely to be affected by baiting programs. This chapter has been published in the journal *Wildlife Research*.

Moseby, K.E., Read, J.L., Galbraith, B., Munro, N., Newport, J and Hill, B.M. (2011). The use of poison baits to control feral cats and red foxes in arid South Australia II. Bait type, placement, lures and non-target uptake. *Wildlife Research* **38**, 350-358.

#### **Chapter 5- Interactions between a top order predator and exotic mesopredators**

The mesopredator release hypothesis predicts that reduced abundance of top-order predators results in increased abundance or activity of smaller predators such as feral cats and foxes and consequently has detrimental impacts on the prey of the smaller predators (Crooks & Soulé 1999). Recent models of decline suggest that dingoes may be an important keystone predator and may facilitate survival of threatened species by suppressing cat and fox abundance (Johnson 2006). This theory was tested experimentally by erecting a 37 square km dingo pen and introducing a pair of dingoes to the pen. Six feral cats and seven foxes from neighbouring pastoral stations were introduced to the pen over a 12 month period. Three control animals of each species were also released into an adjacent unfenced control area without dingoes. The survival of animals in the pen and control areas was compared to determine whether dingoes were able to assist in the control of these feral animal populations. This chapter has been published in the journal, *International Journal of Ecology*.

Moseby, K.E., Neilly, H., Read, J.L. and Crisp, H.A. (2012). Interactions between a top order predator and exotic mesopredators. *International Journal of Ecology* Article ID 250352, 15 pages doi:10.1155/2012/250352.

##### *1.3.2 IMPROVING REINTRODUCTION PROTOCOLS*

#### **Chapter 6- Can predator avoidance training improve reintroduction outcomes for the Bilby (*Macrotis lagotis*) in arid Australia?**

Predator-avoidance training has been used on many taxa in an attempt to improve reintroduction outcomes (Mineka & Cooke, 1988; Magurran, 1989; Mclean *et al.* 1999; Griffin *et al.* 2000). Many prey animals use predator odour to reduce their risk of predation (Perot-Sinal *et al.* 1999; Kats and Dill 1998) often avoiding areas where predator scent is located (Sullivan and Crump 1984). Predator-avoidance training was first tested on greater bilbies within the Arid Recovery Reserve by catching and releasing both trained and untrained animals. Bilbies were captured in nets at night and fitted with radiotransmitters. Control bilbies were released immediately after being fitted with radiotransmitters but trained bilbies were sprayed with cat urine and rubbed with a fresh cat carcass both after capture and upon release in order to associate cats with an unpleasant experience. The behaviour of trained and untrained bilbies was then compared after release including the number of burrows used, distance moved each day, number of burrow entrances etc. Both trained and untrained bilbies were also subjected to two tests between two and three weeks after release. Cat urine was sprayed at their burrow entrance and the soil at the burrow entrance disturbed using a hand trowel to simulate a predator attempting to dig up their prey. The response of the trained and untrained bilbies was compared.

As significant differences in behaviour between trained and untrained bilbies were observed, a release of both trained and untrained bilbies was conducted outside the Reserve into the Wild West zone, an adjacent unfenced area of habitat where cats and foxes were present in low abundance. The behaviour and survival of trained and untrained bilbies post-release was monitored through radiotracking to determine if predator-avoidance training improved survival in the presence of feral cats and foxes. This chapter has been published in the international journal, *Animal Behaviour*.

Moseby, K.E., Cameron, A., and Crisp, H.A. (2012). Can predator avoidance training improve reintroduction outcomes for the Bilby (*Macrotis lagotis*) in arid Australia? *Animal Behaviour* **83**, 1011-1021.

## **Chapter 7- Do release protocols influence translocation outcomes when predation risk is low?**

Reintroduction protocols may have a significant influence on post-release survival, condition and reproductive output of threatened species. Providing food, water or shelter after release may stimulate breeding and/or prevent movement away from the reintroduction site into unprotected areas. Many releases also provide a pen for acclimatisation which has the added advantage of providing initial protection from predator incursions as well as maintaining a protected core population in the long term. However, it is likely that there are interspecific differences in responses to release protocols and that some species may not require soft releases in all circumstances. For example, in areas where introduced predators are completely excluded or where the release site is contained (fenced or island) soft releases may not provide any additional benefit. In order to investigate the differences in post release survival, movement and condition of hard and soft released animals, a comparison of both methods of release was conducted on the greater bilby and burrowing bettong.

Threatened bilbies and burrowing bettongs were re-introduced into the 26 square km northern expansion area of the Arid Recovery Reserve. Half of the released animals of each species were subjected to a soft release and half were hard released directly into the northern expansion area. The soft release animals were placed into an acclimatisation pen in the middle of the northern expansion area and provided with food and water for one month after release. After one month, openings were made in the sides of the netting release pen to allow the

animals to access the northern expansion area at will. The distances moved, survival and shelter sites used by the animals in the two release treatments were compared. A hard release of captive-bred and wild greater stick-nest rats was also conducted into the northern expansion and mortality, movement and shelter site selection compared. Results of this study were used to recommend reintroduction protocols for these species and suggestions for other arid zone mammals. This chapter has been submitted and is currently under review with the international journal, *Biological Conservation*.

Moseby, K.E., Hill, B.M. and Lavery, T. (submitted). Do release protocols influence translocation outcomes when predation risk is low? *Biological Conservation*, under review.

### *1.3.3 PREDICTIVE TOOLS FOR POST-RELEASE MONITORING*

#### **Chapter 8- Keep on counting: The importance of long term post-release monitoring in reintroduction programs**

All four species reintroduced into the Arid Recovery Reserve were monitored for up to 10 years after release. The population size, reproductive rates and body condition of each species were compared with parameters such as season, rainfall, temperature and time since release to highlight trends in post-release population dynamics. Results were used to determine which factors affected post-release abundance and demonstrate the benefits of long term post-release monitoring. The influence of aridity on post-release population dynamics was discussed, and trends used to develop predictions for reintroduction success, propose release protocols and highlight management actions required to ensure population persistence after release. This chapter is being prepared for submission to *Austral Ecology*.

Keep on Counting; the importance of long term post-release monitoring in reintroduction programs.

### *1.3.4 CONCLUSION*

#### **Chapter 9- Improving mammalian reintroduction success in arid Australia.**

In this final chapter I discussed a number of guiding principles designed to improve reintroduction success in arid Australia as well as limitations of my study and suggestions for future research. My results were also discussed in the context of current models of decline in arid zone mammals including the refugia concept, hyper-predation model and the mesopredator release hypothesis. Supported paradigms were used to suggest management actions, reintroduction strategies and research required to improve reintroduction success in arid zone mammals.

## 1.4 STUDY AREA

### 1.4.1 ARID RECOVERY RESERVE

Established in 1997, the Arid Recovery Reserve (30°29'S, 136°53'E) is a 123 square km exclosure situated 20 km north of Roxby Downs in arid South Australia (Fig. 1). A 1.8m high wire netting fence with a curved overhang is used to exclude rabbits, cats and foxes (Moseby and Read 2006). The Reserve is divided into six sections with feral animals sequentially removed from four areas totaling 60 km<sup>2</sup> (Table 1). The second expansion area was kept free of reintroduced species to act as a control area where both introduced and reintroduced species were excluded. The fifth section, the Red Lake Expansion, is currently undergoing feral animal removal and the sixth section, the Dingo Pen (not shown in Fig. 1), is the site of a study involving the interaction between dingoes, cats and foxes.

Red kangaroos (*Macropus rufus*) are present within the Reserve but numbers are controlled by absence of free water and occasional harvesting to maintain low densities more characteristic of pre-European levels. Landsat imagery has demonstrated that plant cover within the reserve has increased relative to outside (Edwards 2001), with the response most evident in perennial grasses (Moseby pers. obs).



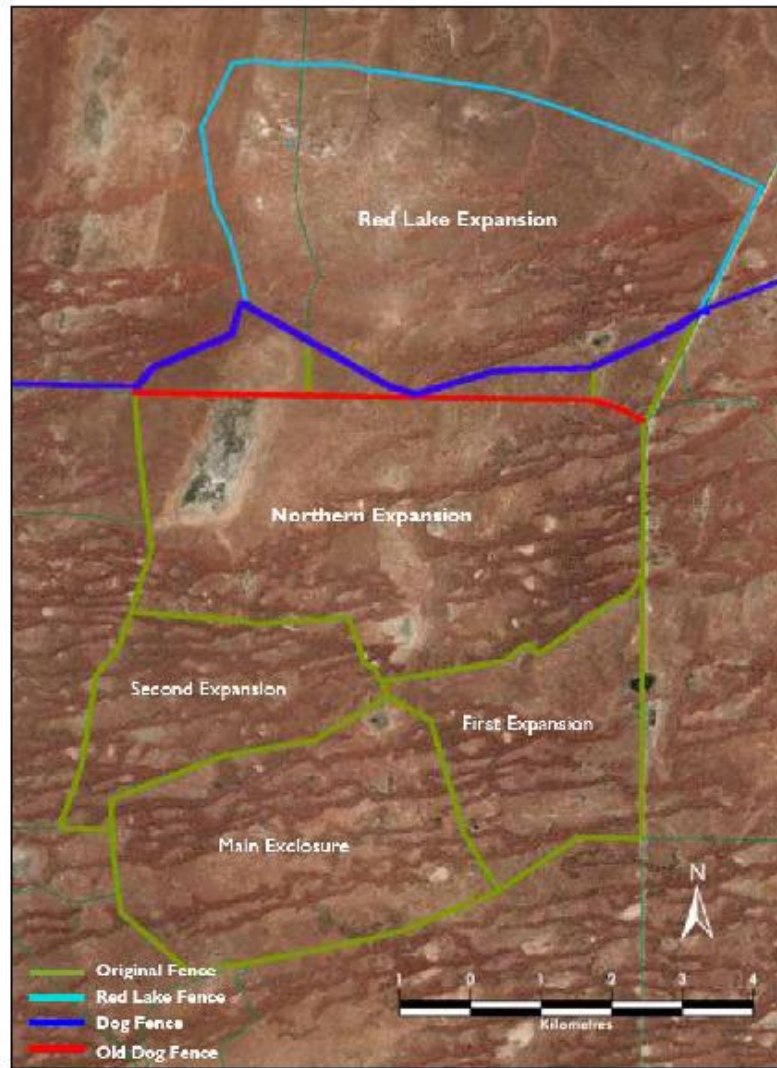


Figure 1: Exclosures within the Arid Recovery Reserve. The dingo pen is located to the north of the Red Lake Expansion but is not shown here. The southern section of the Main Exclosure and First Expansion are also within the Olympic Dam Mine Lease.

### **1.4.1.1 Rainfall History**

The climate of Roxby Down is arid, failing to reach its long term average rainfall of 166 mm in 60 % of years (Read 1995). Rainfall is aseasonal and with equal likelihood during any month. Rainfall during the study period varied considerably (Fig. 2) with above average annual rainfall recorded in 1997, 2001 and 2004 and below average rainfall in 1999 and 2002. Only 44 mm of rain was recorded at the Arid Recovery Reserve in 2002 which was the driest year recorded since the Reserve was established. A long period of drought was experienced from 2006 to 2009 followed by significant rainfall events in 2010.

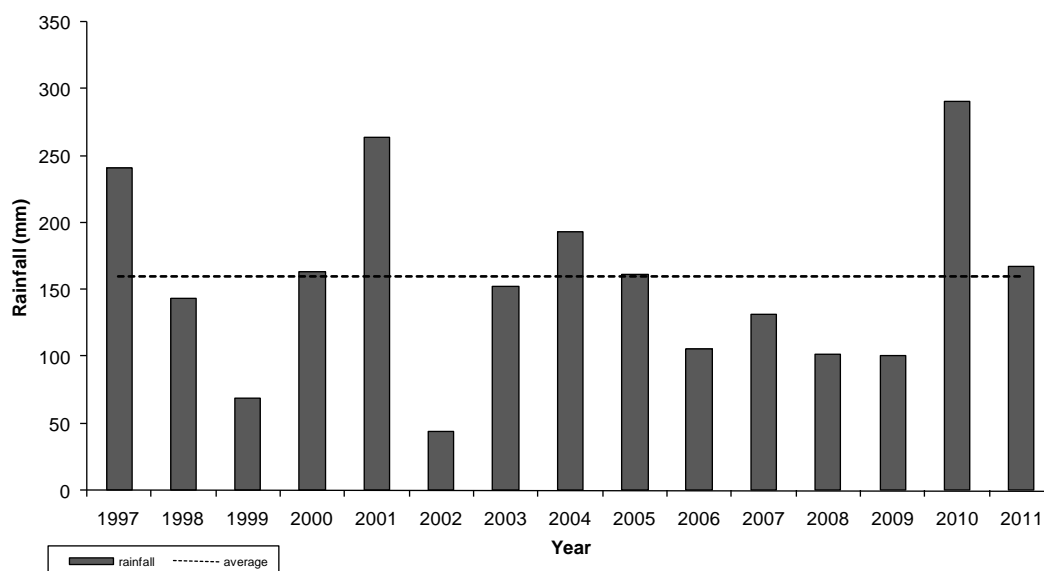


Fig. 2 Rainfall recorded at the Arid Recovery Reserve during the study period. The dotted line indicates the annual average rainfall.

### **1.4.1.2 Habitats**

The dominant landforms within the Reserve are longitudinal orange sand dunes separated by clay interdunal swales. Dunes are generally between 100 m and 1 km apart. Three main habitat types are found within the Reserve; sandhill wattle (*Acacia ligulata*)/hopbush (*Dodonaea viscosa*) dunes, chenopod (bladder saltbush *Atriplex vesicaria*, low bluebush *Maireana astrotricha*) shrubland swales and mulga (*Acacia aneura*) sandplains. Other minor habitat types that are present include native pine (*Callitris glaucophylla*) dunes, canegrass (*Eragrostis australasica*) swamps, claypans and breakaways. Drainage in the Arid Recovery Reserve is endoreic, with water draining into claypans and swamps. In the northern sections, creeklines flow after heavy rain and deposit water into dams and large claypans.

### **1.4.1.3 In situ Fauna**

Regional studies have recorded 48 species of extant reptile and 11 species of native small mammal (Read 1994; BHP Billiton Olympic Dam Operations unpublished data, 2006). Mammals include native rodents such as the spinifex hopping mouse (*Notomys alexis*) which was only re-recorded in the region in 1997 after the release of Rabbit Haemorrhagic Disease (RHD). A study by Moseby *et al.* (2009) found small native rodent abundance, particularly Bolams mouse (*Pseudomys bolami*) and spinifex hopping mouse increased by up to 15 times inside the Reserve compared to outside sites after the removal of rabbits, cats and foxes. The

nationally threatened plains mouse (*Pseudomys australis*), although locally rare, became abundant within the Reserve in 2006 and is now captured each year during annual pitfall and Elliott trapping (Arid Recovery, unpubl. data).

Introduced feral cats, red foxes and European rabbits are present throughout the Roxby Downs region, which is primarily used for cattle (*Bos taurus*) grazing. Domestic stock grazing has occurred in the region for over 100 years and signs of stock damage are evident around artificial and natural waterpoints. Grazing ceased on the Olympic Dam Mining Lease section of the Arid Recovery Reserve in the mid 1980's with the remaining sections lightly grazed until their gradual inclusion into the Reserve between 1999 and 2008.

#### **1.4.1.4 Reintroductions**

A local sub-fossil deposit (Yarra Wurta) was used to determine which species were previously extant in the region (Owens and Read 1999). Locally-extinct mammals were reintroduced to the main enclosure in 1999 (greater stick-nest rat), 2000 (greater bilby) and 2001 (burrowing bettong and western barred bandicoot). Species were then allowed to naturally disperse or were gradually captured and moved to other expansion areas, once feral animals had been removed.

Table 1: Area and species present within each section of the Arid Recovery Reserve.

SNR=greater stick-nest rat, GB=greater bilby, BB=burrowing bettong, WBB=western barred bandicoot. X=absent, √= present.

Section	Size (km <sup>2</sup> )	Reintroduced/native species					Exotic species		
		SNR	GB	BB	WBB	dingo	rabbit	cat	fox
Main enclosure	14	√	√	√	√	X	X	X	X
First expansion	8	√	√	√	√	X	X	X	X
Second expansion	8	X	X	X	X	X	X	X	X
Northern expansion	30	√	√	√	√	X	X	X	X
Red lake expansion	26	X	X	√	X	X	√	√	X
Dingo pen	37	X	X	X	X	√	√	√	√

#### **1.4.1.5 Land use**

The dominant land use in the Roxby Downs area is cattle grazing on pastoral leases. The Arid Recovery Reserve encompasses land from four pastoral stations; Roxby Downs, Stuarts Creek, Mulgaria and Billa Kalina. The Olympic Dam Mine is situated approximately 3km south of the Reserve and produces copper, gold, silver and uranium. Part of the Arid Recovery Reserve is also situated on the Olympic Dam Mine Lease, formerly part of the Roxby Downs Pastoral Station.

The Dog Fence, a vermin proof fence that prevents dingoes from accessing sheep-grazing properties, bisects the study area. Dingoes are controlled to the south of the fence and up to 30 km to the north of the fence with poison meat baits (monosodium fluoroacetate – 1080). The Olympic Dam Mine and Processing Plant are located to the south of the Arid Recovery Reserve.

### 1.4.2 WILD WEST ZONE

The 200 km<sup>2</sup> Wild West Zone (Fig. 3) was established in 2003 adjacent to the Arid Recovery Reserve. This unfenced area of arid zone habitat comprises similar habitats to the Arid Recovery Reserve with slightly more dune habitat. Rabbits, cats and foxes are all found in the Wild West Zone but are controlled through a variety of methods. Initially, control of cats and foxes was attempted using aerial baiting but from 2007 control consisted of a combination of hand baiting, shooting and trapping. Releases of the greater bilby were conducted in 2004 and 2007 and the burrowing bettong was released in 2008. The Wild West Zone is bordered by the Arid Recovery Reserve on the eastern side and the Dog Fence on the northern boundary.

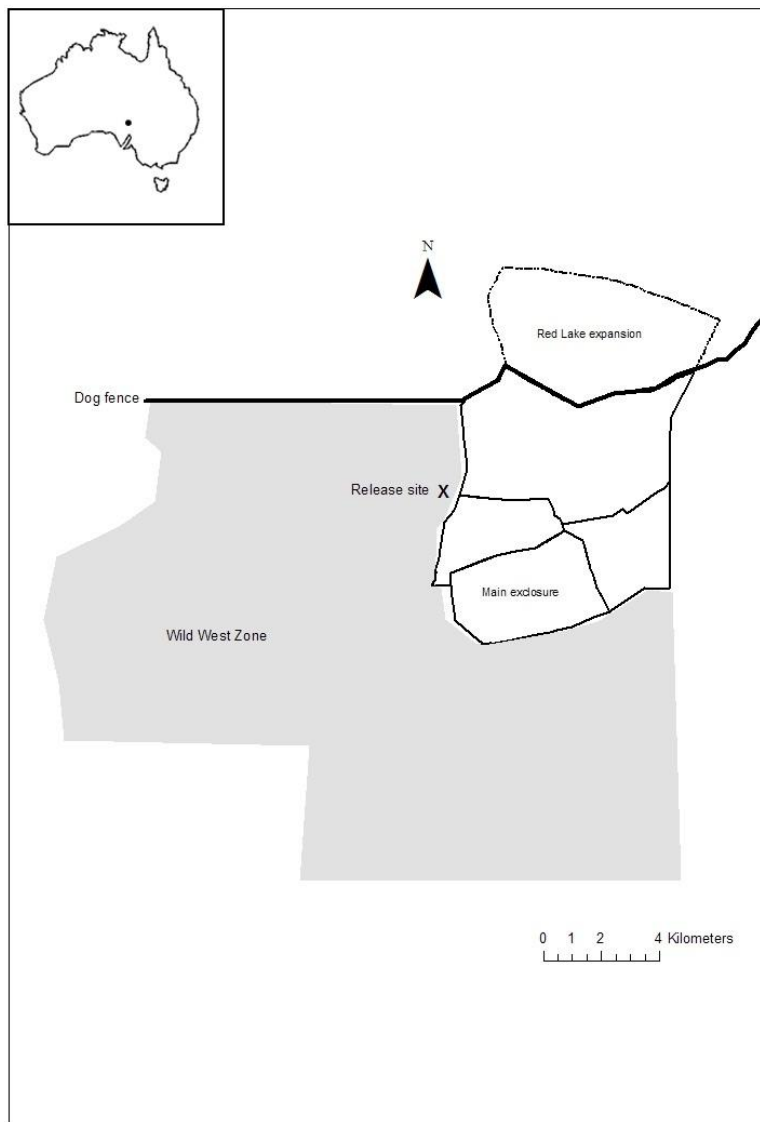


Figure 3: Arid Recovery Reserve showing the unfenced wild west zone and release site for external releases of threatened species.

### 1.4.3 DINGO PEN

The 37 km<sup>2</sup> Dingo Pen was erected in 2008 to the north of the Red Lake Expansion. This pen is surrounded by a 1.6 m high netting fence with a floppy top curving inwards to keep dingoes, cats and foxes within the pen. The pen comprises a number of habitat types including sand dunes, breakaways, creeklines and chenopod stony swales. The southern boundary of the Dingo Pen is a common boundary with the Red Lake Expansion and consists of a lower 1.4 m high fence.

## 1.5 STUDY SPECIES

### 1.5.1 GREATER STICK-NEST RAT (*LEPORILLUS CONDITOR*)

Stick-nest rats are Australian native rodents that were formerly widespread throughout central and southern Australia, mainly within arid and semi-arid regions. Two species have been identified, the greater stick-nest rat and the lesser stick-nest rat (*Leporillus apicalis*). Both species built conspicuous nests under rocky overhangs, within bushes, in caves and in the open. Stick-nest rats were strongly associated with major watershed and drainage areas across the arid and semi arid zone (Copley 1999). Although both stick-nest rat species were often sympatric, it appears the lesser stick-nest rat was restricted to areas with trees in open woodlands whereas the greater stick-nest rat preferred more open shrubland sites without trees (Copley 1999).

Both species became extinct on the mainland during the 1930's (Copley 1999). South Australian mainland specimens of the greater stick-nest rat have been collected only from the western shore of lake Eyre (Finlayson 1941) and the Nullabor Plain but subfossil material reveals a much wider former distribution (Copley 1999). Natural extant populations are now found only on East and West Franklin Islands in South Australia, discovered by Professor Wood Jones in 1920 (Jones 1924). Since then they have been successfully re-introduced to St Peters and Reevesby Island in S.A., Salutation Island in W.A. and the Arid Recovery Reserve in northern South Australia. Other mainland reintroduction trials have occurred at Yookamurra, Scotia and Heirisson Prong but have failed most likely due to lack of cover and/or predation from native and exotic predators.

The greater stick-nest rat is thought to be almost entirely herbivorous but the lesser stick-nest rat also included some insects in its diet (Copley 1999). The most common dietary items for the greater stick-nest rat were the leaves and fruits of chenopods (Bolam 1927, Finlayson 1941, Copley 1988). Its former distribution was defined mainly by perennial semi-succulent and succulent shrubs (Copley 1999). Ryan *et al.* (2003) found some dietary overlap with the European rabbit but the greater stick-nest rat preferred more succulent chenopod species whilst rabbits preferred shrub seedlings and grasses.

The nests are made from sticks and plant material and can house pairs or colonies of individuals. Nests are often woven within and around a small bush (pers. obs.). The size of nests is described by explorers and early naturalists as between 0.6 m cm and 1.5 m in height and 1-2 m in diameter (Copley 1999). Nests in caves and under overhangs have been reported up to 5m across and 2m high although most were 1.5-2m across and 0.5-1m high

(Copley 1999). In the Murray Darling Basin, early explorers Sturt and Browne described stick nest rat nests as housing between 6 and 10 rats and often being 5 foot high (Browne 1844-55 in Finnis 1966). Copley (1999) also describes a mounted constable in 1896 who noted 4-5 rats in each nest but this was probably a nest of the lesser rather than greater stick-nest rat. Franklin Island rat nests are usually occupied by individuals, male-female pairs or females with young and sometimes young from a previous litter. On Reevesby Island, rat nests contain single rats, male-female pairs or females with young from the previous 1-3 litters. If older young were present they were almost always female (Copley 1999).

On the Franklin Islands, stick-nest rats breed all year round and have 1-2 young (Copley 1988) although in captivity, litter size can occasionally be 3 or even 4. Young are attached to the teats initially and when older are left in the nest. Gestation is approximately 44 days and weaning is at 30-40 days (Copley 1988).

Native predators include barn owls (*Tyto alba*), goannas (*Varanus spp.*) and snakes, although snakes and goannas appear to prey mainly on subadult rats (Schwaner unpublished in Copley 1999; Bolton and Moseby 2004). Captive bred greater stick-nest rats released to the wild have been known to be preyed on by feral cats, foxes, Goulds goannas (*Varanus gouldii*), barn owls and Australasian boobooks (*Ninox novaeseelandiae*) (Pedler and Copley 1993; Bolton and Moseby 2004).

Re-introduced female greater stick-nest rats have small stable home ranges and males have larger less defined ones (Pedler and Copley 1993). Copley (1999) considers suitable cover and habitat containing succulent and semi-succulent perennials to be key requirements of suitable reintroduction sites. He considers the absence of introduced herbivores and presence of cover for protection from predators (native and introduced) key requirements.

### 1.5.2 GREATER BILBY (*MACROTIS LAGOTIS*)

The greater bilby is a burrowing, nocturnal bandicoot that digs extensively for the seeds, bulbs and invertebrates that constitute its diet. The greater bilby has declined since European settlement and although still found in the arid interior of Northern Territory, Western Australia and Queensland, the species now occupies less than 20 % of its former range (Southgate 1995). The greater bilby was formerly the most widespread of the South Australian bandicoots (Kemper 1990). Jones (1924) considered the greater bilby to be extant in the Lake Eyre Basin and Ooldea at the time of his writings and the last South Australian museum records were obtained in the far North West of the state and the Lake Eyre Basin in 1933 (Kemper 1990). Greater bilby remains were present in the Yarra Wurta Subfossil deposit found 30 km north of Roxby Downs in the early 1990's suggesting they were formerly present in the study region. The greater bilby is now listed as Vulnerable under the Federal Environmental Protection and Biodiversity Conservation Act (1999).

The greater bilby has been successfully re-introduced to some mesic areas and islands within its former range where cats and foxes are absent, intensively controlled or eradicated (e.g. Thistle Island and Yookamurra Sanctuary in S.A.). Reintroductions into inland arid zone areas have proved more difficult due to the problems of controlling and excluding introduced predators from a large area without the use of peninsulas or costly fences. A trial reintroduction into Watarrka National Park in the Northern Territory failed due to high levels of predation (Southgate pers comm.) but reintroductions into fenced arid zone reserves such

as Arid Recovery in northern South Australia and Currawinya in SW Queensland have proved more successful.

### 1.5.3 BURROWING BETTONG (*BETTONGIA LESUEUR*)

The burrowing bettong or boodie, is the only member of the macropod family to construct and live in underground warrens. The burrowing bettong once had one of the largest geographic ranges of any Australian mammal (Burbidge 1995) but by 1993 the species occupied less than 0.01% of its former range (Short and Turner 1993). The burrowing bettong is now extinct on the mainland with the last museum specimen obtained in 1942 (Kitchener and Vicker 1981). The burrowing bettong is now found naturally on only three islands off the coast of Western Australia; Bernier, Dorre and Barrow Islands. Populations on the islands appear stable and the species has been successfully reintroduced to the mainland at several predator-free sites including Arid Recovery, Scotia Sanctuary and Heirisson Prong (Short and Turner 2000). The burrowing bettong is nationally listed as Vulnerable (EPBC Act 1999).

Bettongs are nocturnal and feed on a variety of foods including roots, foliage, seeds and fruits (Robley *et al.* 2001). Although present, hypogeal fungi do not feature as prominently in its diet as that of related rat kangaroos (Taylor 1992; Robley *et al.* 2001), possibly due to the fact that burrowing bettongs inhabit drier regions. Bettongs breed continuously when conditions are favourable and are capable of producing up to 3 young per year (Tyndale-Biscoe 1968). A decline in breeding in late spring and early summer has been reported in some re-introduced populations (Short and Turner 2000). Pouch life is approximately 115 days and dominant males establish a group of females within warren systems. Sexual maturity is attained at approximately 280 days (Tyndale-Biscoe 1968). Warrens may have numerous entrances and one warren on Barrow Island was found to have 120 entrances (Burbidge 1995), but smaller warrens supporting 20-40 individuals are more common (Sander *et al.* 1997). Burrows are often dug in sandy or calcrete soils and contain social units usually made up of one male and several females (Sander *et al.* 1997). Bettongs have a polygynous mating system and trapping studies generally capture more males than females (Short and Turner 1999). Animals remain relatively close to their warrens and prefer to feed in sandy habitats such as dunes and sandplains (Short and Turner 1999; Finlayson and Moseby 2004).

There is little dietary overlap with rabbits suggesting that rabbits alone are unlikely to have caused the mainland extinction of bettongs (Robley *et al.* 2001). Christensen and Burrows (1994) suggested that burrowing bettongs were highly vulnerable to fox and cat predation due to their gregarious behaviour, colonial lifestyle, slow moving gait and regularity of movements between warrens and feeding grounds. Short (1998) concluded that the extinction of burrowing bettongs in New South Wales was caused by the red fox. Short and Turner (2000) attributed their successful reintroduction of Burrowing Bettongs to Heirisson Prong to adequate control of foxes and cats, however the population has recently crashed significantly due to a combination of drought and cat incursions (J. Short pers. comm.). Bettong numbers on Bernier and Dorre Island fluctuate depending on drought conditions (Short and Turner 1999).

Burrowing bettongs dig warrens for shelter, but also regularly dig for seeds and roots producing foraging digs of various depths and shapes. Both warrens and foraging digs affect soil characteristics such as carbon levels and water infiltration and lead to increased germination of seedlings (James and Eldridge 2007; James *et al.* 2009). Their role as ecosystem engineers has been recently highlighted and suggests that their demise since



European settlement has had broader implications for soil and vegetation health (Noble *et al.* 2007).

#### 1.5.4 WESTERN BARRED BANDICOOT (*PERAMELES BOUGAINVILLE*)

The western barred bandicoot formerly had a wide distribution in southern and western Australia (Kemper 1990) with most records obtained from the arid and semi arid regions. In South Australia, most records are south of a line running across the top of Lake Torrens. The last confirmed South Australian museum record was in 1922 from Ooldea (Kemper 1990) and in Western Australia from Rawlinna in 1929 (Kitchener and Vicker 1981). Subfossil material was collected from the Yarra Wurta subfossil deposit in the 1990's from about 30km north of Roxby Downs. Western barred bandicoots are likely to have occurred over a wide range of habitats in arid and semi arid areas. Gould (1863 in Short *et al.* 1998) reported them in dense scrub in Western Australia, while in South Australia and New South Wales they were found in stony ranges and vast open plains (Short *et al.* 1998). *Perameles* may have been found in more open habitats than *Isoodon* bandicoots, possibly contributing to their extinction on the mainland due to increased vulnerability to introduced predators.

Three subspecies have been recognised, the South Australian subspecies *P. bougainville notina* and eastern mainland species, *P. bougainville fasciata*, are now extinct and the only remaining subspecies, *P. bougainville bougainville*, is found naturally only on Bernier and Dorre Islands in Western Australia (Friend 1990). Western barred bandicoots were reintroduced to the Arid Recovery Reserve in 2001, and in 2012 were present throughout the 60 km<sup>2</sup> feral-free areas of the Arid Recovery Reserve.

Western barred bandicoots are the smallest member of the bandicoot family (*Peramelidae*) and weigh about 250g. They are nocturnal and nest during the day in a sandy depression made under leaf litter. Western barred bandicoots are primarily insectivorous but they also consume other foods such as seeds, roots and plant material (Richards and Short 2003). Jones (1924) found a definite breeding season in western barred bandicoots in South Australia during May and June. In extant populations in Western Australia, more young are produced in the wetter winter months on Dorre and Bernier Islands (Short *et al.* 1998) where 70% of the annual rainfall falls between May and August. The onset of breeding appears to be triggered by the first substantial rains in autumn, following summer drought (Short *et al.* 1998). Although breeding is linked to regular winter and spring rainfall it may also be opportunistic during years of summer rainfall (Short *et al.* 1998). The average annual rainfall on Bernier and Dorre Islands is 252 mm compared to only 160 mm at Roxby Downs. Richards and Short (2003) found breeding throughout the year with a winter peak in a reintroduced population on Heirisson Prong.

Adult sex ratios are male biased (Short *et al.* 1998). Western barred bandicoots have fewer pouch young per litter than other species of bandicoots, the young are smaller, and spend less time in the pouch before being left in the nest (Short *et al.* 1998). The pouch life is 45-60 days. The oldest surviving female bandicoot born on Heirisson Prong was at least 4 years and 3 months old and had pouch young. Western barred bandicoots are solitary animals (Short *et al.* 1998) and only females share a nest with their young.

Recent identification of a wart-like papillomavirus in the wild island populations has caused concern for the future of western barred bandicoot populations. Ocular chlamydiosis has also

been identified from island bandicoots leading to corneal opacity, conjunctivitis and ocular discharge (Warren *et al.* 2005).

Between 1988 and 1991, Dorre island bandicoots exhibited a rate of increase of 0.645 and doubling time of 1.07 years (6 fold increase in 3 years). But on Bernier Island the population increased only slightly. No seasonal pattern of trap success was evident. In a reintroduced population, Richards and Short (2003) found the rate of increase of Known To Be Alive (KTBA) bandicoots over 4 years to be 0.54 after release with a doubling time of 1.3 years. Populations decline significantly in response to drought (Short *et al.* 1998).

#### 1.5.5 FERAL CAT (*FELIS CATUS*) RED FOX (*VULPES VULPES*) AND EUROPEAN RABBIT (*ORYCTOLAGUS CUNICULUS*)

Control of the feral cat (*Felis catus*) and red fox (*Vulpes vulpes*) is often the most important management action required for successful re-establishment of threatened mammal species in Australia (Kinnear *et al.* 2002). The introduced red fox is also considered a key threat to many threatened species in North America (Lewis *et al.* 1999) and feral cats have been implicated in the extinction of several species in Mexico (Wood *et al.* 2002). The red fox has been successfully controlled in many areas using poison meat baits but feral cat control through poison baiting has often been less effective due to poor bait uptake (Risbey *et al.* 1997; Kinnear *et al.* 1998; Burrows *et al.* 2003; Algar and Burrows 2004; Hegglin *et al.* 2004; Olsson *et al.* 2005; Algar *et al.* 2007).

Within Australia, feral cat home range sizes of between 0.29 and 22.06 km<sup>2</sup> have been reported (Jones and Coman 1982; Edwards *et al.* 2001; Burrows *et al.* 2003; Molsher *et al.* 2005). Although few studies have been conducted in the arid zone, home-range and movements of arid zone feral cats appear to be much larger than in other environments (Moseby *et al.* 2009; Edwards *et al.* 2001; Burrows *et al.* 2003), perhaps reflecting the lower productivity.

The first fox was recorded near the study region in 1905 at Anna Creek Station. It is not known when feral cats first became established in the region but populations are now resident in all habitat types and are not reliant on local towns or rubbish dumps for food or shelter. Feral cat and fox densities in the study region fluctuate according to seasonal conditions and averaged ~0.8 and 0.6 per km<sup>2</sup> respectively, during a 10-year period prior to my study (Read and Bowen 2001). Regional targeted control is limited to irregular shooting from amateur shooters.

The European rabbit was introduced to Australia in 1860 and quickly spread across the continent. By 1910, rabbits had populated nearly all of Australia south of the Tropic of Capricorn (Johnson 2006) and were first recorded in the study region in 1895. Rabbits cause widespread vegetation damage and can reach plague proportions after significant rainfall events.

Since its arrival, the European rabbit has caused extensive damage to plants in arid regions and is likely to have significantly changed the vegetation composition and abundance. Rabbits feed on a variety of plants preferring grasses and herbaceous species. Rabbit density declined significantly in the arid zone after the introduction of Rabbit Haemorrhagic Disease in 1996 but recent data suggest numbers are beginning to increase due to increased immunity in the population. Before the introduction of RHD in 1995, rabbit density averaged between

100 and 150 per km<sup>2</sup> (Fig. 4, Bowen and Read 1998). Rabbit densities are estimated by spotlight counts, and after the introduction of RHD they averaged between 0 and 10 per km<sup>2</sup> from 1996 to 2000, gradually increasing to an average of approximately 50 per km<sup>2</sup> from 2004 to present (Fig. 4, BHP Environmental Department, unpubl. data).

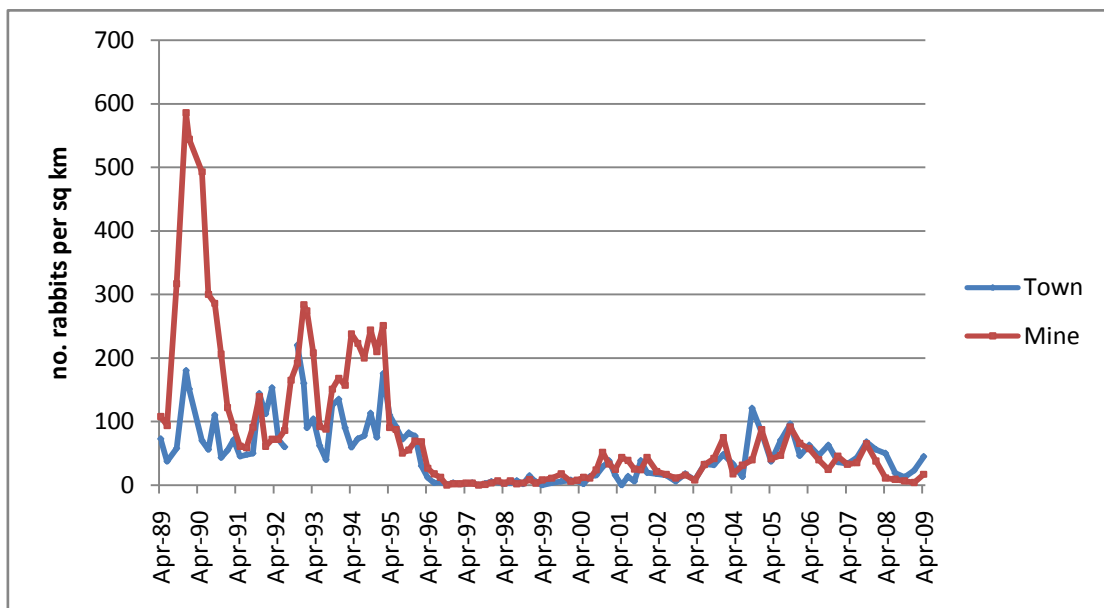


Fig .4: Abundance of rabbits per km<sup>2</sup> estimated using spotlight counts at Roxby Downs township (town) and Olympic Dam Mine (mine). Data courtesy of BHP Billiton, Environment Department.

## 2.CHAPTER TWO: PREDATION DETERMINES THE OUTCOME OF 10 REINTRODUCTION ATTEMPTS IN ARID SOUTH AUSTRALIA.

Published as a research article in the journal Biological Conservation;

Moseby, K.E., Read, J.L., Paton, D.C., Copley, P., Hill, B.M. and Crisp, H.M. (2011).  
Predation determines the outcome of 11 reintroduction attempts in arid Australia. *Biological Conservation* **144**, 2863-2872.

Chapter 2- Moseby, K.F., Read, J.L., Paton, D.C., Copley, P., Hill, B.M. and Crisp, H.A. (2011). Predation determines the outcome of 11 reintroduction attempts in arid Australia. *Biological Conservation* 144, 2863-2872.

**Author Contribution**

Concepted and designed the experiments- KEM, JLI, PC, DCP

Performed the experiments including field work- KEM, JLR, HAC, BWH

Analysed the data- KEM

Contributed reagents/materials/analysis tools

Wrote the paper- KEM

**Author Declaration**

I approve the above author contribution statement and give permission for the paper/s co-authored by me to be included in the thesis by Katherine Moseby

---

Name: John L. Read

Signature:

Date: \_\_\_\_\_

Name: Dr D.C. Paton

Signature:

Date: \_\_\_\_\_

Name: P. Copley

Signature:

Date: \_\_\_\_\_

Name: B.M. Hill

Signature:

Date: 7

Name: H.A. Cris

Signature:

Date: 7/3/10

## Abstract

Ten reintroduction attempts were conducted in and around the Arid Recovery Reserve in northern South Australia between 1998 and 2008. Five locally-extinct mammal species and one reptile species were reintroduced into a fenced Reserve where cats, foxes and rabbits were excluded. Reintroductions of the nationally threatened greater stick-nest rat, burrowing bettong, greater bilby and western barred bandicoot were all considered successful based on short and medium-term success criteria. These criteria included continued survival after eight years, increased distribution across the large reserve and, most importantly, recovery after a drought event. The trial reintroductions of the numbat and woma python into the reserve were unsuccessful due to predation by native avian and reptilian predators respectively. Outside the Reserve, where cats and foxes were present but controlled through poison baiting, reintroduction attempts of the greater bilby and burrowing bettong were unsuccessful. High mortality was attributed to cat and fox predation with dingoes also contributing to post-release mortality in bettongs. However, a reintroduction of burrowing bettongs into a fenced area with low rabbit and cat abundance has, to-date, met short-term and medium-term success criteria. Results suggest that the absence or severe restriction of exotic mammalian predators was the critical factor responsible for the success of the mammal reintroductions. Determining thresholds of predator activity below which successful reintroduction of threatened species can occur, are needed to improve the science of reintroduction biology in Australia.

## Key Words

reintroduction, translocation, threatened species, success criteria, arid zone

## 2.1 INTRODUCTION

Reintroduction programs are a tool often used in species recovery programs, both in Australia (Short *et al.*, 1992; Christensen and Burrows, 1994; Gibson *et al.*, 1994; Southgate and Possingham, 1995; Priddel and Wheeler, 1997; Priddel and Wheeler, 2002) and overseas (Wolf *et al.*, 1998; Seddon *et al.*, 2007). However, most reintroductions of rare or threatened species fail to establish viable populations (Griffith *et al.*, 1989; Dodd and Seigel, 1991; Beck *et al.*, 1994) and the majority of mammal reintroductions onto mainland Australia have also failed (Short *et al.*, 1992; Gibson *et al.*, 1994; Fischer and Lindenmayer, 2000; Short, 2009). Australian practitioners typically attribute failures to predation by exotic predators such as cats (*Felis catus*) and foxes (*Vulpes vulpes*) (Short, 2009), with successful reintroductions occurring on islands or into exclosures where introduced predators are absent (e.g. Richards and Short, 2003). However, studies comparing reintroduction success in adjacent areas with and without exotic predators are necessary to determine the role of predation relative to death from disease, starvation, hyper-dispersal and stress, particularly when scavenging by predators could mask the cause of death.

The Australian arid zone mammal fauna (Finlayson, 1941, 1961) and their habitats have been severely altered since European settlement through overgrazing by rabbits (*Oryctolagus cuniculus*) and domestic stock, increased predation from cats and foxes, an increase in

artificial watering points and changes to vegetation composition (Friedel 1985; Morton, 1990; Wilson, 1990). Translocations typically have a poor chance of success unless habitat quality is high (Griffith *et al.*, 1989) but contemporary vegetation condition may not be adequate to support reintroduced herbivorous animals. The few Australian mainland arid zone reintroductions that have occurred have focused on insectivorous species such as bandicoots and bilbies (Christensen and Burrows, 1994; Southgate, 1994; Moseby and O'Donnell, 2003), revealing little about the ability of herbivorous species to re-establish in diminished quality habitat.

Another factor that may thwart reintroduction attempts in the arid zone is the lack of arid-adapted source populations. Droughts and temperature extremes are a feature of arid environments and may have exerted considerable selective pressure on mammal populations. Many former arid zone mammal subspecies such as the western barred bandicoot, (*Perameles bougainville notina*) are now extinct leaving only the more mesic island-dwelling subspecies such as *Perameles bougainville bougainville* available for reintroduction.

Although the primary aim of any reintroduction is to establish a self-sustaining (Griffith *et al.*, 1989), free-ranging viable population (IUCN, 1998; Fischer and Lindenmayer, 2000), Bajomi (2010) sourced 12 definitions of reintroduction success from the global literature. The four common definitions were: using Population Viability Analysis to predict whether the population will be self-sustaining (Beck *et al.*, 1994); population persistence (Wolf *et al.*, 1998); breeding of the first wild-born generation (Sarrazin and Barbault, 1996; Seddon, 1999); and a positive population growth rate over three generations, or ten years (IUCN, 1998). Many Australian practitioners also set *a priori* criteria for reintroduction success (Backhouse *et al.*, 1994, Short and Turner, 2000; Richards and Short, 2003; Vale *et al.*, 2004) such as a percentage survival after 12 months, reproduction within 6 months and population persistence after five years. These criteria allow short-term success to be measured through targeted monitoring but may have little bearing on long-term success (see Soorae, 2010).

Ten reintroductions of locally-extinct species were attempted in and around the fenced Arid Recovery Reserve in northern South Australia between 1998 and 2008. Various source populations and release sizes were used at three release locations with no, low or moderate levels of introduced rabbits, cats and foxes. This study outlines the success or failure of each reintroduction and investigates factors affecting reintroduction outcomes. We identified short, medium and long-term success criteria with only the first two stages considered here.

## 2.2 METHODS

### 2.2.1 STUDY SITES

Established in 1997, the Arid Recovery Reserve (30°29'S, 136°53'E) is a privately owned 123 km<sup>2</sup> fenced exclosure situated 20 km north of Roxby Downs in arid South Australia (Fig. 1). Reintroductions were attempted into two paddocks of the Reserve (Fig. 1); the Main Exclosure where cats, rabbits and foxes were eradicated, and the Red Lake Paddock where low levels of rabbits and cats were present and contained. Additional reintroductions were attempted into the adjacent unfenced Wild West Zone where cats and foxes were free-ranging but subjected to ongoing control measures.

The Main Exclosure is a 14 km<sup>2</sup> paddock where rabbits, cats and foxes were eradicated in 1999 (Read *et al.*, 2011b). An estimated 1000 rabbits (Moseby and Read, 2006) and 6 cats and foxes were removed during eradication. A 1.8 m high wire netting fence with a curved overhang and two electric wires was used to exclude rabbits, cats and foxes (Moseby and Read, 2006). Reintroductions of native fauna occurred either into a small 10 ha release pen within the Main Exclosure or directly into the Main Exclosure itself.

The 26 km<sup>2</sup> Red Lake Paddock supports low levels of rabbits and cats. It is surrounded by a 1.15 m high fence with a curved overhang that excludes rabbits, foxes and most cats (Moseby and Read, 2006; K. Moseby pers. obs.). More than 4000 rabbits and three cats have been removed over a five year period through trapping and poisoning (Read *et al.*, 2011b), although both species are yet to be eliminated. No foxes were present in the pen during the study.

The 200 km<sup>2</sup> unfenced Wild West Zone is bordered by the Arid Recovery Reserve to the east and the dog fence to the north (Fig. 1). The dog fence is a man-made wire netting fence designed to exclude dingoes (*Canis lupus dingo*) from southern sheep grazing areas. Only the Arid Recovery fence-line formed a significant barrier to rabbits, cats and foxes. Rabbit, cat and fox abundance was higher than that recorded in the Red Lake Paddock but was limited by ongoing control. Cats and foxes were controlled in the Wild West Zone through annual (2002-2004) then quarterly (2005-2006) aerial baiting using Eradicat<sup>TM</sup> sausage baits (Western Australian Department for Environment and Conservation) or dried meat baits both containing 1080 (monofluoroacetate) poison. From 2007, control took the form of bimonthly Eradicat<sup>TM</sup> ground baiting at a density of 10 to 25 per km<sup>2</sup>, opportunistic poison baiting in areas where feral cat tracks were observed, weekly shooting and permanent trapping at up to ten sites using soft-catch foothold traps. Rabbit control was not attempted.



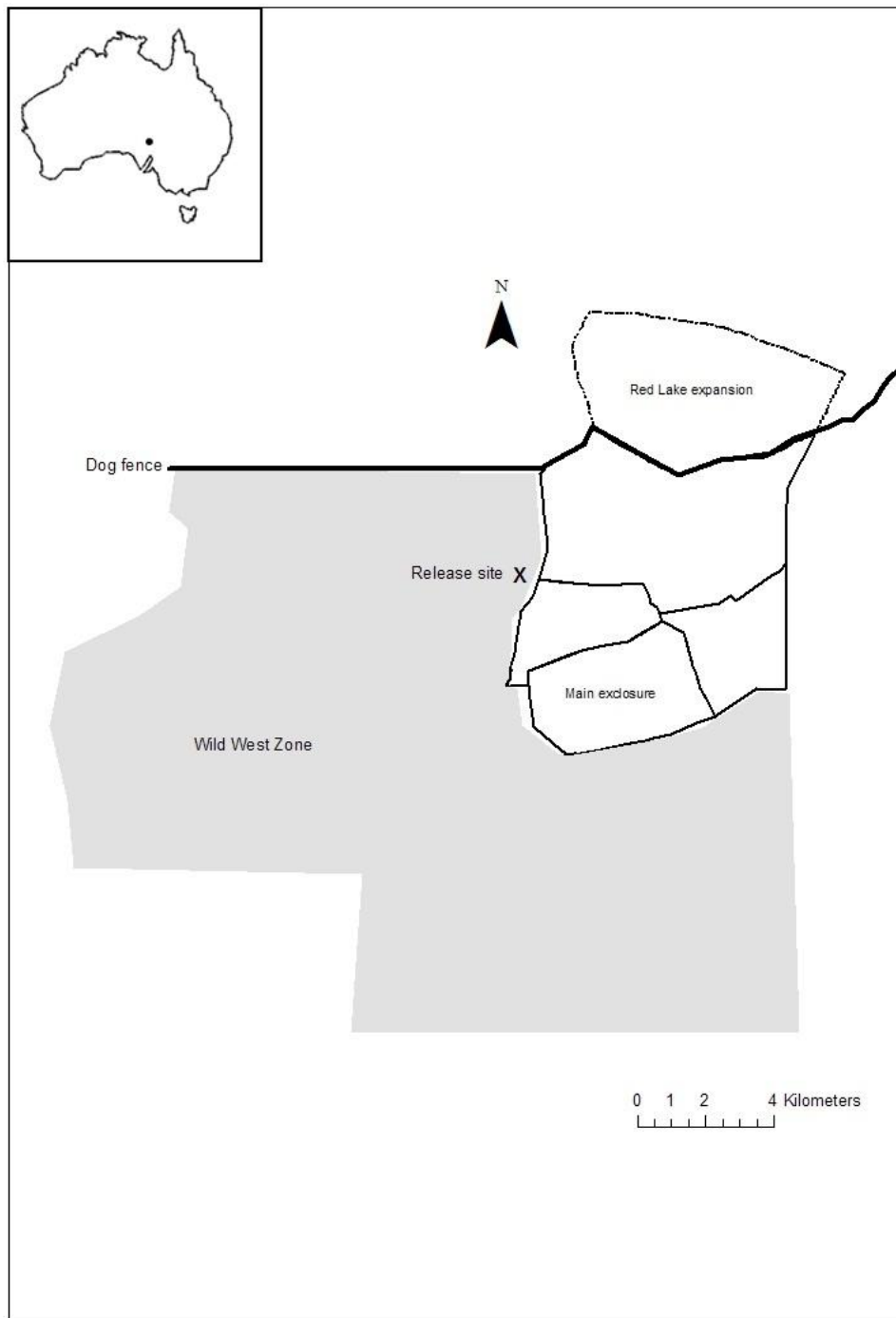


Fig. 1: Map of reserve showing areas of reintroductions (Main Enclosure, Red Lake and Wild West Zone), the dog fence and the release site within the Wild West Zone.

The Roxby Downs climate is arid, failing to reach its long term average rainfall of 166 mm in 60 % of years (Read, 1995). Rainfall during the study period varied considerably with above average annual rainfall recorded in 1997 (240 mm), 2001 (263 mm) and 2004 (193 mm) and

very dry conditions in 1999 (69 mm) and 2002 (44 mm). Rainfall in other years varied between 100 and 160 mm.

The dominant landforms within the three release areas are longitudinal orange sand dunes separated by clay interdunal swales. Dunes are generally spaced 100 m to 1 km apart. Three main habitat types are present; sandhill wattle (*Acacia ligulata*)/hopbush (*Dodonaea viscosa*) dunes, chenopod (*Atriplex vesicaria*)/(*Maireana astrotricha*) shrubland swales, and mulga (*Acacia aneura*) sandplains. Drainage is endoreic, into claypans and swamps.

The Roxby Downs region, has historically been used for sheep (*Ovis aries*) and cattle (*Bos taurus*) grazing. Stock damage is focused around artificial and natural waterpoints. Grazing ceased on the southern section of the Main Exclosure in the mid 1980's and the remaining sections of the Reserve were lightly grazed with negligible detectable influence on extant reptile and mammal capture rates (Read and Cunningham, 2010) until their gradual inclusion into the Reserve between 1999 and 2008.

### 2.2.2 REINTRODUCTIONS

Museum records, literature, subfossil deposits (Owens and Read, 1999) and old nests and burrows were used to compile an inventory of the pre-European vertebrate fauna of the Roxby Downs region. Locally-extinct species were selected for reintroductions based on their previous distribution in the area and their availability. Translocation proposals were prepared and implemented for each species by Arid Recovery, a partnership between BHP Billiton, The University of Adelaide, the local community and The South Australian Department for Environment and Natural Resources (DENR). Proposals were approved by the DENR and where source populations were located in Western Australia, permission and export permits were also obtained from the Western Australian Department for Environment and Conservation.

Five IUCN-listed mammal species; the greater stick-nest rat (*Leporillus conditor*), the greater bilby (*Macrotis lagotis*), the burrowing bettong (*Bettongia lesueur*), the western barred bandicoot (*Perameles bougainville*) and the numbat (*Myrmecobius fasciatus*); and one IUCN-listed reptile species, the woma python (*Aspidites ramsayi*) were reintroduced to the Main Exclosure between 1998 and 2001 (Table 1). Due to the limited numbers available for release, the greater bilbies and western barred bandicoots were released into a 10 ha release pen within the Main Exclosure where they were provided with supplementary food and water. After a few months, dispersal holes were cut in the sides of the release pen and animals allowed free access to the rest of the Main Exclosure. The numbats, although also constituting a small release size, were released directly into the Main Exclosure as they are termite specialists and roam over large areas to forage (Friend, 2008). Stick-nest rat and burrowing bettong releases occurred in two stages, a small trial release into the release pen with food and water provided followed by a full-scale release into the Main Exclosure a year later. Source populations were preferentially obtained from wild stocks (Table 1). Captive-bred individuals were descendants of wild individuals captured from the Franklin Islands (greater stick-nest rat) or Strzelecki Desert (woma python) in South Australia, or deserts in Western Australia and the Northern Territory (greater bilby). More detailed information on reintroduction protocols is available for the numbat (Bester and Rusten, 2009), woma python (Read *et al.*, 2011a) and greater bilby (Moseby and O'Donnell, 2003).

One burrowing bettong reintroduction was attempted into the Red Lake Paddock in 2008 and involved a direct transfer of wild animals from the Main Exclosure without any supplementary food or water (Table 1). Two greater bilby and one burrowing bettong reintroduction attempts were made into the Wild West Zone between 2004 and 2008 (Table 1). Release animals were sourced from within the Reserve and released in a single location (Fig 1) with no supplementary food or water provided.

### *2.2.3 MONITORING*

Both native and feral species were monitored using track counts at all three release locations. Within the Main Exclosure and the Red Lake Paddock, track counts involved walking along longitudinal sand dunes and recording animal tracks that crossed a 1 m wide path. To ensure consistency, an animal's track was counted each time it entered and left the path. Between 2000 and 2010, the number of tracks of greater stick-nest rat, greater bilby, burrowing bettong, western barred bandicoot and numbat were recorded along 8 x 1 km segments within the Main Exclosure between September and December each year. The presence or absence of tracks in each segment was also used to determine trajectories of distribution change within the Reserve since release. Between 2000 and 2005, the track transects were conducted on the morning after a windy day and a still night, to ensure only fresh tracks from a single night were counted. After this time the population of reintroduced species increased to the degree that it became necessary to clear old tracks the day prior to tracking by dragging a 1 m bar and chain behind a quadbike. Approximately 11 km of transect, divided into 3 x 3-4 km lengths, was also sampled in the Red Lake Paddock using this method from 2008 until 2010.

A subsample of all reintroduced species released into the Main Exclosure and Wild West Zone was monitored using radiotracking for at least one month after release (Table 1). After approximately one month, radiocollared animals were recaptured and weighed to determine weight loss. Cage trapping also occurred annually within the Main Exclosure and Red Lake Paddock to determine reproductive condition and record the presence of second generation individuals. Trapping was not used for density comparisons due to interspecific differences in trapability. Reintroduced species in the Wild West Zone were monitored using a combination of radiotracking, burrow monitoring and track counts along 50 km of dune crests on a quad bike. Predator activity was also noted along these threatened species track transects which were located within 7 km of the release site (Fig. 1) and conducted monthly from January 2008 to March 2009.

Table 1: The characteristics of reintroduced populations and their source populations. Two trial releases and 10 full-scale releases were conducted. Reintroductions in bold were deemed successful.

<i>Species</i>	<i>Year</i>	<i>Location of Source Population</i>	<i>Wild / Captive bred</i>	<i>Climate of Source Population</i>	<i>Release Location</i>	<i>Release Pen</i>	<i>Release size(M.F)</i>	<i>No. Radio-tracked</i>
<b>Greater stick-nest rat<sup>T</sup></b>	<b>1998</b>	<b>Reevesby Island, S.A.</b>	<b>Wild</b>	<b>Mesic</b>	<b>Arid Recovery Reserve</b>	<b>Yes</b>	<b>8(4.4)</b>	<b>8</b>
Greater stick-nest rat	1999	Reevesby Is/Monarto Zoo, S.A	Wild/Captive Bred	Mesic	Arid Recovery Reserve	No	92/6^(55.43)	30/6
<b>Burrowing bettong<sup>T</sup></b>	<b>1999</b>	<b>Heirisson Prong, W.A.</b>	<b>Wild</b>	<b>Semi-arid coastal</b>	<b>Arid Recovery Reserve</b>	<b>Yes</b>	<b>10(3.7)</b>	<b>10</b>
Burrowing bettong	2000	Bernier Island, W.A.	Wild	Semi-arid coastal	Arid Recovery Reserve	No	20(8.12)	10
<b>Greater bilby</b>	<b>2000</b>	<b>Monarto Zoo, S.A.</b>	<b>Captive Bred</b>	<b>Mesic*</b>	<b>Arid Recovery Reserve</b>	<b>Yes</b>	<b>9(4.5)</b>	<b>9</b>
<b>Western barred bandicoot</b>	<b>2001</b>	<b>Bernier Island, W.A.</b>	<b>Wild</b>	<b>Semi-arid coastal</b>	<b>Arid Recovery Reserve</b>	<b>Yes</b>	<b>12(2.10)^</b>	<b>11</b>
Numbat	2005	Scotia Sanctuary, N.S.W.	Wild	Semi-arid inland	Arid Recovery Reserve	No	5(3.2)	5
Woma Python	2007	Zoos S.A.	Captive Bred	Mesic*	Arid Recovery Reserve	No	9(7.2)	9
Greater bilby	2004	Arid Recovery	Wild	Arid	Wild West Zone	No	12(8.6)	12
Greater bilby	2007	Arid Recovery	Wild	Arid	Wild West Zone	No	20(7.13)	20
Burrowing bettong	2008	Arid Recovery	Wild	Arid	Wild West Zone	No	101(58.43)	15
<b>Burrowing bettong</b>	<b>2008</b>	<b>Arid Recovery</b>	<b>Wild</b>	<b>Arid</b>	<b>Red Lake Paddock</b>	<b>No</b>	<b>67</b>	<b>0</b>

<sup>T</sup>=trial release

\*= original source stock from the arid deserts, captive breeding facilities located in mesic area near Adelaide

^= One female animal died before it could be released

(M.F.) = sex ratio of released animals

Regional feral animal activity within the Wild West Zone was monitored using 45 short tracking segments which were sampled from 2003 to 2009. Segments were a modified version of the monitoring technique established by Engeman and Allen (2000) and each comprised a 200 m length of vehicle track on sandy substrate, separated by a distance of at least 500 m. Dingo tracks on successive segments were counted only once as the spacing was designed to be optimal for cat monitoring and dingoes often follow roads for several kilometres (Read and Eldridge, 2010). Segments were driven over by four wheel drive vehicle in the late afternoon on the day preceding sampling to obliterate older tracks and the following morning an observer walked each segment and recorded the presence or absence of fresh cat, fox, rabbit and dingo tracks on the vehicle track. Segments were pooled to determine the percentage with cat, fox and dingo presence and compared over time. Unfortunately this method could not be duplicated in the Red Lake Paddock due to insufficient lengths of vehicle tracks for adequate replication. Instead, feral animal tracks were counted along the 11 km of walking track transects used for monitoring burrowing bettongs.

#### 2.2.4 SUCCESS CRITERIA

We considered short, medium and long-term success criteria for reintroduction success (Table 2). Time lines for long-term success have not yet been met so only short and medium-term criteria are addressed here. An important medium-term criterion was the persistence and recovery of the population through drought, a common occurrence in the Australian arid zone that can cause widespread decline or local extinction in some mammal species (see Brandle and Moseby, 1999; Masters, 1993; Moseby *et al.*, 2006). A drought is defined as a prolonged, abnormally dry period and is classified according to rainfall deficiencies over a certain period (Australian Bureau of Meteorology, 2011). Severe rainfall deficiency periods of 12 months or more, when rainfall was among the lowest 5 % on record, were used to determine the start and end of drought conditions at Olympic Dam (10-15 km from the reintroduction sites) between 1998 and 2010.

Sutherland *et al.* (2010) outlined recommendations for monitoring and documenting bird reintroductions. Where appropriate, we also recorded these details for our mammal and reptile reintroductions (Table 3).

Table 2: Short, medium and long term criteria used to determine reintroduction success.

<i>Criteria</i>	<i>Details</i>
Short <sup>1</sup>	Survival of more than 50% of radiocollared individuals after 1 month
Short <sup>2</sup>	weight loss less than 15% of body weight after 1 month
Short <sup>3</sup>	Independent young produced within 12 months
Medium <sup>1</sup>	Second generation produced within 2 years
Medium <sup>2</sup>	Increased distribution after 5 years
Medium <sup>3</sup>	Population extant after 8 years
Medium <sup>4</sup>	Population recovery after a drought event
Long <sup>1</sup>	Genetic diversity maintained
Long <sup>2</sup>	No significant loss of carrying capacity through intraspecific habitat alteration
Long <sup>3</sup>	Minimal intervention required for population regulation

Table 3: The fate of 10 reintroduction events and 2 trial releases inside and outside the Arid Recovery Reserve including the presence or absence of feral species.

<i>Species</i>	<i>Year</i>	<i>Feral Species Present</i>			<i>Short Term Success Criteria</i>				<i>Medium Term Success Criteria</i>			<i>Success</i>	<i>Reason for Failure</i>
		Cat	Fox	Rabbit	50% survival	<15% Weight loss	Indep. young	2 <sup>nd</sup> gen	distribution	Extant 8 years	Drought recovery		
Greater stick-nest rat <sup>†</sup>	1998	x	x	x	Yes	Yes	Yes	-	-	-	-	-	
Greater stick-nest rat	1999	x	x	x	Yes	No	Yes	Yes	Yes	Yes	Yes	YES	
Burrowing bettong <sup>†</sup>	1999	x	x	x	Yes	Yes	Yes	-	-	-	-	-	
Burrowing bettong	2000	x	x	x	Yes	Yes	Yes	Yes	Yes	Yes	Yes	YES	
Greater bilby	2000	x	x	x	Yes	No	Yes	Yes	Yes	Yes	Yes	YES	
Western b bandicoot	2001	x	x	x	Yes	Yes	Yes	Yes	Yes	Yes	Yes	YES	
Numbat	2005	x	x	x	Yes	No*	No	No	No	No	-	NO	Bird of Prey predation
Woma Python	2007	x	x	x	No	No*	No	No	No	No	-	NO	Mulga snake predation
Greater bilby	2004	√	√	√	No	Yes	Yes	No	No	No	-	NO	Cat predation
Greater bilby	2007	√	√	√	Yes	Yes	Yes	No	No	No	No	NO	Predation+drought
Burrowing bettong	2008	√	√	√	Yes	No*	No	No*	No	No	No	NO	Predation
Burrowing bettong	2008	√	x	√	Yes	-	Yes	Yes	3 to date	3 to date	Yes	YES	

\*= animals lost more than 15% of body weight after 3 months rather than 1 month

<sup>†</sup> =these reintroductions were trial releases with full scale releases occurring within 12 months. Thus the medium term success criteria cannot be accurately determined for these trial release populations as their offspring cannot be distinguished from the full scale release.

## 2.3 RESULTS

Two drought periods were identified in the 12 years between 1998 and 2010 based on severe rainfall deficiencies, a 12 month deficiency from January 2002 to January 2003 when only 43 mm of rain were recorded for the period (average of 3.3 mm per month) and an 18 month deficiency from May 2007 to October 2008 when 74 mm of rain was recorded (average of 4 mm per month).

### 2.3.1 MAIN EXCLOSURE REINTRODUCTIONS

No cats or foxes were detected in the Main Exclosure during this study. The releases of 106 stick-nest rats, 30 burrowing bettongs, nine greater bilbies and 12 western barred bandicoots were all successful and populations fluctuated over time but generally increased (Moseby and O'Donnell, 2003; Moseby and Bice, 2004; Fig. 2). Only one of 157 translocated animals died in transit (a wild sourced stick-nest rat) and one female western barred bandicoot died during a vet check just prior to release. The only deaths in the first few weeks after release were two captive-bred and four wild sourced stick-nest rats. All these deaths were thought to have been due to stress-related trauma. A minimum of two of the three short-term criteria were met with significant weight loss recorded initially in some species but more than 50 % of released animals survived the first month and reproduction occurred within 12 months (Table 3). One male bilby and one female stick-nest rat both lost more than 15 % of body weight but went on to regain the weight after four to five months. When medium-term criteria were considered, second generation individuals were produced and all species increased their distribution within the Arid Recovery Reserve in the five years after release (Fig. 3). Stick-nest rat, burrowing bettong and greater bilby tracks were recorded on more than 75 % of track transect segments within one year of release (Fig. 3) and their nests and burrows were observed in all segments within three years. Western barred bandicoots took longer to colonise the Reserve but were present in nearly 90 % of segments within four years.

All four species met the third medium-term reintroduction criterion of being extant within the Reserve eight years after release (Table 3). Population recovery after drought, the important fourth success criterion, was tested during the droughts of 2002 and 2008. Track counts of bilby and stick-nest rat declined by 50 % during the 2002 drought and both western barred bandicoot and bilby tracks declined by more than 50 % during the 2008 drought. All populations recovered and began increasing within two years of the droughts breaking (Fig. 2). Burrowing bettong tracks did not decline during either drought event.

The numbat and woma python releases into the Arid Recovery Reserve were both unsuccessful with reintroduced animals killed by native predators. The numbat release met no medium-term success criteria and only one of the short-term criterion, namely 50 % of released animals were still alive one month after release. Of the five numbats released, one male and two females were killed by bird of prey at 47 days and seven months after release respectively (Bester and Rusten, 2009). The other two male numbats, including one that temporarily lost more than 25 % of its body weight after release, were recorded on track transects up until three years after release. All of the reintroduced woma pythons were killed within four months, with mulga snake (*Pseudechis australis*) predation confirmed or implied in all cases (Read *et al.*, 2011a). This predation occurred despite raising the pythons to five years of age (mean snout-vent length 1,312 ±SE 58 mm, mean weight 1,032 ±SE 140 g)



before release, so that they were only slightly smaller than the mulga snakes that killed them. No short-term success criteria were met, more than 50 % were dead within one month of release, no breeding occurred and significant weight loss was recorded. We were not able to recapture woma pythons after one month as they were sheltering in burrowing bettong warrens but when two snakes were opportunistically recaptured at 77 days and 112 days after release they had lost 24 % and 27 % of their body weight respectively. However, none of the pythons appeared emaciated or unhealthy whenever they were observed directly during the trial and successful feeding was confirmed for one python, which contained substantial fat bodies during necropsy following its death 101 days after release (Read *et al.*, 2011a).

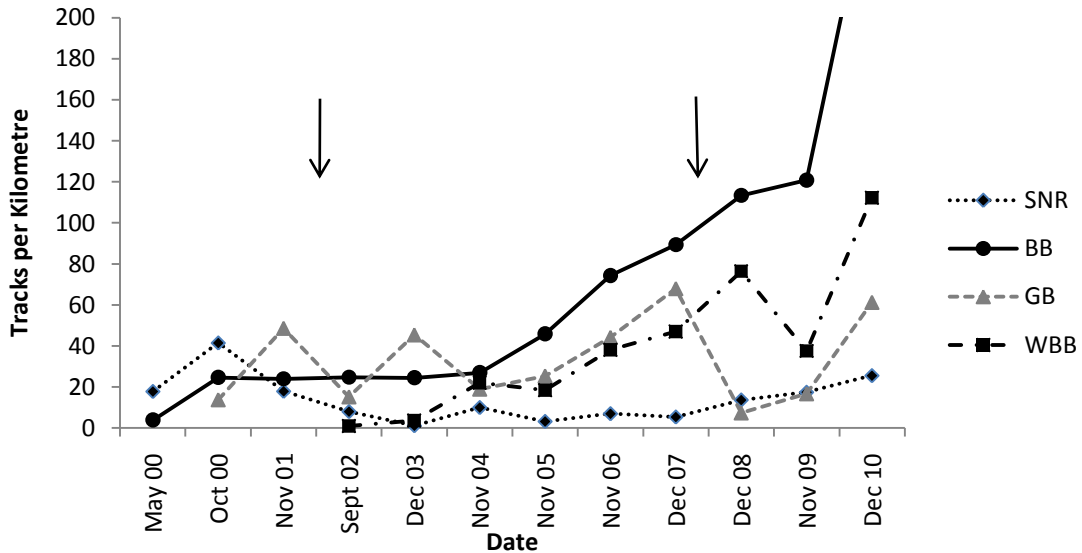


Fig. 2: Annual activity counts for species reintroduced into the Main Enclosure of Arid Recovery. The total length of transects is 8 km. December 2010 bettong track counts exceeded 200 but the axis scale was reduced to show trends for other species. Arrows indicate drought events.

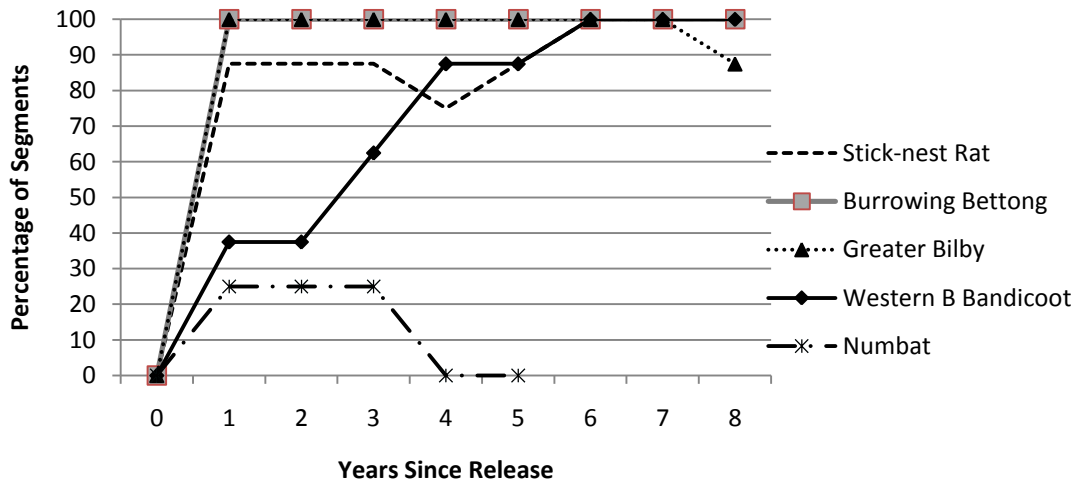


Fig 3: The increase in distribution of reintroduced species in the Main Enclosure of the Arid Recovery Reserve after release. Distribution is shown as the percentage of track transect segments with tracks present. Total transect segments is 8 (8 x 1 km).

### 2.3.2 RED LAKE REINTRODUCTION

No foxes were recorded within the Red Lake Paddock during the study and cat activity ranged from zero to one track intercept per kilometre during 2008 and 2009, increasing to three per kilometre in 2010 (Fig. 4). Rabbit track activity was low with less than five tracks per kilometre recorded (Fig. 4). The release of 67 burrowing bettongs in September 2008 was deemed successful based on both short-term and medium-term success criteria (Table 3). Track monitoring indicated that activity remained stable after release suggesting that less than 50 % of individuals died in the first month and the first success criterion was met (Fig. 4). Track activity mirrored that recorded inside the Main Enclosure in the first few years after release, with activity gradually increasing (Figs. 2 and 4). Trapping one year after release recorded untagged animals that had been recruited into the population, several with their own pouch young. Bettong tracks and burrows were recorded on all three track transect segments within one year of release indicating that they had increased their distribution and met the third medium-term success criterion. Although animals were released during the drought of 2008, the bettong population increased after the drought, thereby satisfying the last medium-term success criterion.

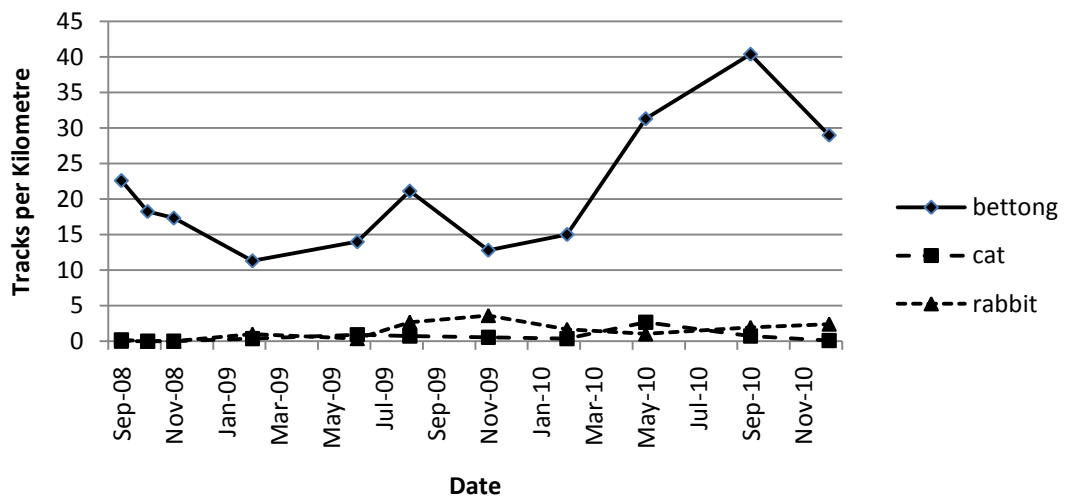


Fig. 4: Track activity of burrowing bettongs, rabbits and feral cats within the Red Lake Paddock area after bettongs were released in September 2008. Total transect length is 11 km.

### 2.3.3 WILD WEST REINTRODUCTIONS

All three reintroduction attempts into the unfenced Wild West Zone were unsuccessful despite meeting some short-term success criteria. During the first greater bilby release in June 2004, seven out of the 12 bilbies died within 25 days of release, contravening the first short-term success criterion (Table 3). Predation was the only confirmed cause of death with six deaths from feral cats and one from an unknown predator. Cat tracks were observed around fresh carcasses, some cats were captured at cached remains, and one cat trapped in the area had bilby remains in its stomach. The other two short-term criteria were met, four bilbies that were still alive three months after release had maintained or increased their weight and at least three juvenile bilbies were known to survive to pouch exit. Another adult bilby was killed by a cat eight months after release and a population of bilbies outside the Reserve failed to

become established. Bilbies exhibited high site fidelity with all females and most males remaining within 2 km of the release site. Cat and fox tracks were recorded on between 15 and 30 % of regional track segments in the Wild West Zone over the period of the reintroduction (Fig. 5) and cat tracks were commonly observed in the release area. Dingo activity was recorded on less than 10 % of track segments and tracks were never observed within 4 km of the release site.

During the second Wild West Zone bilby release in August 2007, bilbies exhibited high site fidelity and persisted for longer. All three short-term criteria were met and only two of the 20 released bilbies died in the first month, one from suspected malnutrition and one from predation by a wedge-tailed eagle. A cat killed a third bilby at four months post-release. Tracks of the released adult population of bilbies remained relatively stable in the Wild West Zone for 12 months after release until July 2008 when there was a significant decline (Fig. 6). Juvenile bilby tracks were first recorded along track transects five months after release and peaked in May 2008 after which time no juvenile tracks were recorded (Fig. 6). Juvenile tracks were followed to burrows where fresh cat tracks were always observed around the entrance. The last adult bilby track was observed 19 months after initial release in January 2009 and the release did not meet any medium-term success criteria (Table 3). All females and four of the seven male bilbies remained within 4 km of the release site and frequent cat and occasional fox tracks were recorded during monthly threatened species track counts in this area. On the regional feral track segments, cat activity was three times higher than fox or dingo during the first five months after release (25 to 35 % of segments) before declining to 15 % in July 2008 (Fig 5). There was a simultaneous decline in bilby, rabbit and predator activity that corresponded with an intensification of drought conditions suggesting increased predation pressure from a declining cat and fox population under nutritional stress. Despite their regional occurrence (Fig. 5), dingoes were not implicated in the deaths of any bilbies as they were not recorded within 4 km of the release site on monthly threatened species track counts until November 2008, several months after the bilby decline.

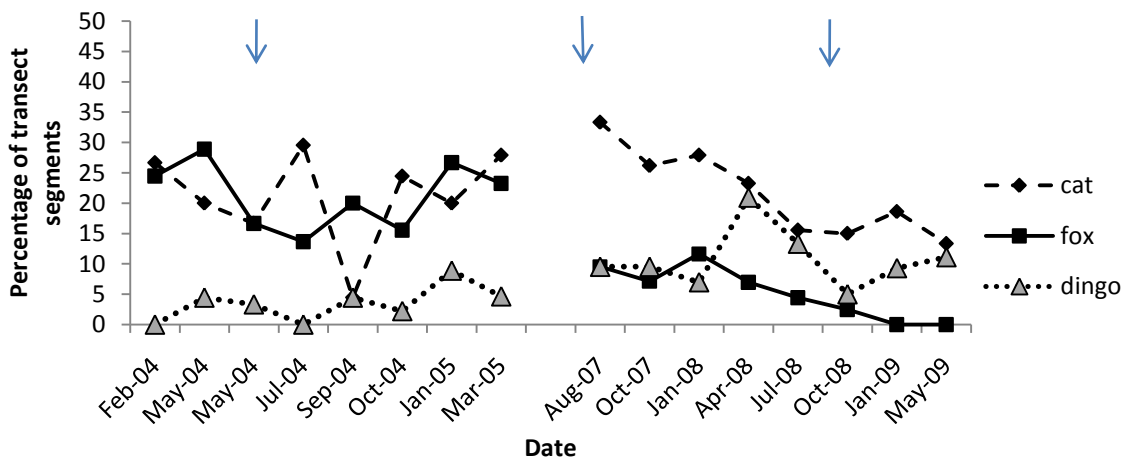


Fig. 5: Predator activity outside the Reserve in the Wild West Zone between 2004 and 2009. Arrows indicate timing of Wild West reintroduction attempts. Total number of segments is 45.

One hundred and one burrowing bettongs were released into the Wild West Zone in September 2008. Fourteen of the fifteen radiocollared bettongs died within four months of release and no animals were known to be alive at seven months post-release (March 2009). Only one short-term success criterion was met, less than 50 % mortality in the first month.

Some recaptured bettongs had lost more than 15 % of their body weight but this weight loss was also recorded inside the Reserve during the same period. Small pouch young were observed at three months post-release but no independent young were recorded. Fifty four rabbit warrens within the Wild West Zone showed signs of bettong occupancy at one month after release, declining to zero at seven months post-release. Seven cat, two dingo and one fox track were observed at monitored warrens. Four radiocollared bettongs and ten un-collared bettongs were killed at the release site within 24 hours of release. Track identification and post mortem analysis by Zoos S.A. revealed that the cause of death in all these cases was a dingo or wild dog.

Six of the 15 radiocollared bettongs left the Wild West Zone and travelled up to 18 km away into areas where cats and foxes were not controlled. When retrieved, their bodies were too decomposed for autopsy but predation was considered the cause of death because radiocollars showed signs of predator damage; misshapen metal collar bands, teeth puncture marks in the rubber surrounding the metal collar and/or fur stuck to the band. These long distance movements out of the Wild West Zone are likely to have permitted increased predation rates probably from cats and foxes, as dingoes are rarely recorded south of the dingo fence where the dispersed bettongs were found. Cat tracks were present on two to three times more track segments than both fox and dingo tracks during the first four months after release (Fig. 2).

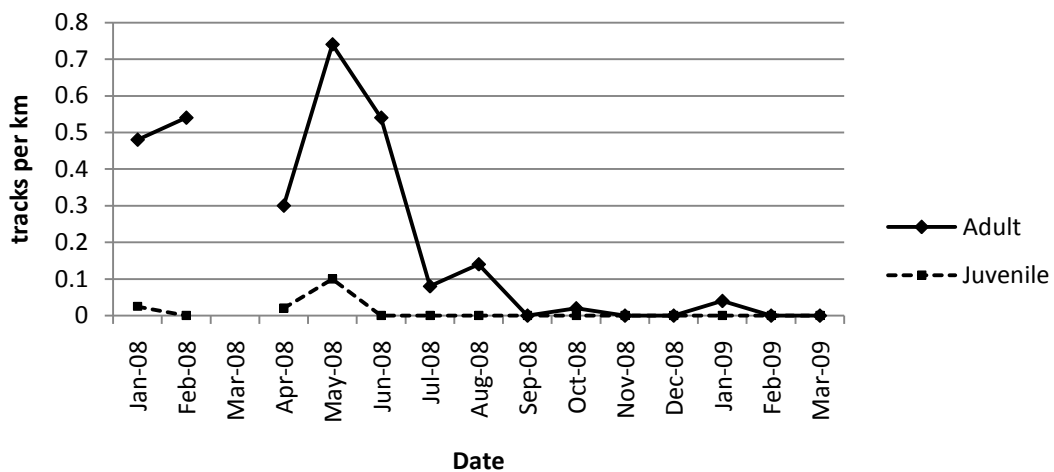


Fig. 6: Track activity of reintroduced greater bilbies in the Wild West Zone after their release in August 2007. Track monitoring did not begin until January 2008 after tail radiotransmitters were removed. Total transect length is 50 km.

## 2.4 DISCUSSION

Half of the ten reintroduction events were unsuccessful with predation the key factor responsible for reintroduction failure. Native predators were responsible for two failures inside the Reserve and cat activity levels outside the Reserve were apparently too high for

successful reintroductions of bilbies in this area. Predation from cats, foxes and dingoes was thought to be responsible for the failure of the burrowing bettong release outside the Reserve. Predation has been responsible for the failure of many global reintroduction programs (Fischer and Lindenmayer, 2000), and 80 % of mammal reintroduction failures in Australia are attributed to predation (Short, 2009). Other greater bilby reintroductions have been successful in Australia only where cats and foxes have been excluded, with failed reintroduction attempts into unfenced arid areas also attributed to predation (Christensen and Burrows, 1994; Southgate, 1994; Southgate and Possingham, 1995).

Reintroduced greater bilbies survived significantly longer than burrowing bettongs outside the Reserve in the presence of exotic predators. Elsewhere, Southgate and Possingham (1995) also found some adult bilby individuals alive 28 months after reintroduction into Watarrka National Park in northern Australia where introduced predators were present. These results, coupled with our observation that juvenile bilbies were not surviving, suggest that whilst adult bilbies may tolerate some level of feral predator presence, juveniles may be particularly vulnerable to cat predation. Other studies suggest that cats may impact threatened species such as burrowing bettongs (Short and Turner 2000) and black flanked rock wallabies (Read and Ward 2010) through their predation on juveniles.

There is some evidence that drought may be a particularly vulnerable time for reintroduced populations when exotic predators are present. Reintroduced populations of the greater bilby and burrowing bettong both inside and outside the Reserve suffered from population decline or weight loss during the drought in 2008, but only outside populations became extinct. The dry conditions and subsequent decline in rabbit activity may have precipitated prey switching, resulting in increased predation pressure on reintroduced populations. Drought also likely lowered reproductive rates, increasing the susceptibility of the population to predator-driven extinction.

Interestingly, the bettong reintroduction into the Red Lake Paddock in the presence of very few cats has met most medium-term success criteria. The absence of foxes in this paddock is likely to have contributed to reintroduction success and evidence from other studies suggest that in some cases, only a proportion of cats are able, or inclined, to be significant predators of certain prey species (Spencer, 1991). The reintroduced western barred bandicoot at Heirisson Prong in Western Australia was able to successfully establish in the presence of one cat per 4-6 km<sup>2</sup> but at significantly lower densities than cat-free islands (Richards and Short, 2003). Future monitoring will determine whether bettong densities reach those recorded in predator-free areas of the Reserve.

The killing of 17 burrowing bettongs by a single dingo on the night of release outside the Reserve implies that surplus killing by dingoes may also represent a threat to some reintroductions. The first few hours after release are arguably the time when predator-naive animals are most susceptible to surplus killing and it is not known what impact the dingo may have had on an established population.

Improved control of exotic predators, particularly cats, is needed before broadscale reintroductions into unfenced arid areas are likely to be successful. Determining thresholds of predator activity below which successful reintroduction of threatened species can occur are urgently needed to improve the science of reintroduction biology in Australia. Predator thresholds are likely to vary according to species, season, habitat and alternative prey abundance and so require a national experimental approach including standards for measuring terrestrial predator abundance. This would enable results from reintroduction attempts to be more widely comparable and allow for increased hypothesis testing in future releases.

Thresholds, together with the development of broadscale predator control methods, should form the basis for threatened species recovery in Australia over the next decade.

Apart from predation, other factors had no obvious impact on reintroduction success. Our findings supported work by Short (2009) who found no correlation between reintroduction success and source populations in Australia in comparison to Fischer and Lindenmayer (2000) who reported higher reintroduction success by global practitioners who used wild stock. We also used a one way ANOVA to compare release sizes in successful and unsuccessful reintroductions but found no significant difference. The majority of bird and mammal reintroductions in Australia and overseas between 1973 and 1986 comprised less than 75 animals (Griffith *et al.*, 1989) and whilst many studies have found larger release groups more successful (Griffith *et al.* 1989; Wolf *et al.*, 1998; Fischer and Lindenmayer, 2000), smaller groups have had a higher success rate in Australia (Short, 2009). However, the covariate influence of introduced predators on the success of small or captive-bred release groups has not been investigated. The majority of successful Australian mammal reintroductions with small or captive-bred founders have been into predator-free environments (see Richards and Short, 2003; Finlayson *et al.*, 2008; present study). When exotic predators are present, large release groups (see Sinclair *et al.* 1998) and wild source populations may be imperative. In our study, the relatively large release size of 67 bettongs into the Red Lake Paddock where cats were present may have been integral in securing the success of this release.

The low abundance of rabbits in the Red Lake Paddock and absence of rabbits in the Main Exclosure may have contributed to the success of reintroductions in these areas. Reintroductions into areas where rabbits are present have been successful (Richards and Short, 2003; Currawinya Reserve Queensland, Peter McRae pers. comm.) but removing or controlling exotic predators can lead to significant increases in rabbit numbers (Newsome *et al.*, 1989; Robley *et al.*, 2002). Rabbits can cause severe vegetation damage (Lange and Graham, 1983) and Richards and Short (2003) found litter size in reintroduced western barred bandicoots increased with a decrease in rabbit abundance. However, Robley *et al.* (2002) found no influence of rabbits on the survival, recruitment or rate of increase of reintroduced burrowing bettongs. The main threat to reintroduced species from rabbits appears to be the secondary influence of sustaining higher predator numbers which in turn prey on native species, especially as rabbit numbers decline in drought conditions.

The use of a release pen did not improve medium-term reintroduction success within the Reserve but may have assisted with the welfare of individuals during the early stages of release. Release pens may be more useful in areas where predators are present as they allow competitors and both exotic and native predators to be removed (Richards and Short, 2003). In large release areas, pens may also be useful for preventing hyperdispersal after release (Short and Turner, 2000), a problem encountered with our external bettong release and in some other reintroduction programs (Christensen and Burrows, 1994; Short and Turner, 2000; Flinders Ranges National Park, Peter Copley pers. obs). Males that disperse large distances from the release site are unlikely to contribute to the population and may be at higher risk of predation (Steen, 1994; Norrdahl and Korpimaki, 1998; Anthony and Bloomstein, 2000).

Reintroducing species from non-arid areas did not appear to affect the success of reintroductions suggesting that local extinction of arid-adapted populations may not be an impediment to re-establishment in arid areas. Poor habitat quality has hampered other reintroduction attempts (Griffith *et al.*, 1989; Wolf *et al.*, 1998) but despite high historical rabbit abundance in the Arid Recovery region (Bowen and Read, 1998), habitat quality appeared sufficient to support breeding populations of two primarily herbivorous species, the stick-nest rat and burrowing bettong.

The failure of the woma reintroduction through predation by mulga snakes was not predicted prior to release; woma pythons are known predators of elapid snakes, they were nearly as large as adult mulga snakes when released and high densities of mulga snakes had not been recorded within the Reserve. This failed reintroduction attempt reinforces the benefits of closely monitoring releases, particularly when the ecology, behaviour and abundance of the reintroduced species and extant *in situ* species are not thoroughly understood.

In general, short-term criteria such as weight loss and production of independent young were not accurate predictors of medium-term reintroduction success with nine out of ten releases meeting at least one short-term success criterion but only five meeting medium-term criteria (Table 3). Short term criteria may be useful as early health indicators for triggering management interventions and provide initial insight into habitat quality and the success of any predator control activities. In arid areas, medium-term criteria should include the ability to recover from drought events as these are times of low reproductive output when species are arguably most vulnerable to predation and local extinction. Criteria referring to pre-defined population increases are less relevant since many arid zone species exhibit large fluctuations in abundance depending on seasonal conditions (Newsome and Corbett, 1975; Dickman *et al.*, 1999).

Seddon (1999) states that the ultimate objective of any reintroduction is “population persistence without intervention” but concedes that this is unrealistic in many cases. Exclosure fences require ongoing maintenance, predator incursions must be addressed and monitoring should continue indefinitely. We have attempted to minimise the need for intervention by increasing the size of the Arid Recovery Reserve to 123 km<sup>2</sup> in order to accommodate large populations. Larger reserve area and population size will improve genetic viability, possibly allow low levels of predator incursions to be tolerated and increase the chance of intercepting patchy rainfall events to reduce drought impacts. Other long-term success criteria include no significant loss of carrying capacity through intraspecific habitat alteration, a criterion developed after burrowing bettongs reached high densities and began impacting on vegetation within the Reserve. Ideally, the reintroduced population should not inadvertently cause their own decline through actions such as overbrowsing. Continued attempts to reintroduce native predators will hopefully lead to a self-sustaining ecosystem where minimal intervention is required.

## 2.5 ACKNOWLEDGEMENTS

Arid Recovery is a conservation initiative supported by BHP Billiton, The University of Adelaide, The South Australian Department for Environment and Natural Resources and the local community. Funding for this project was generously provided by Nature Foundation SA, WWF Threatened Species Network, The Australian Federal Government’s Natural Heritage Trust and Envirofund, BHP Billiton, the Wildlife Conservation Fund and Australian Geographic. Other organisations such as Zoos S.A, Western Australian Department for Environment and Conservation and the Australian Wildlife Conservancy provided considerable in kind support. I am indebted to the many Arid Recovery staff and volunteers who assisted with the project including Adam Bester, Jackie Bice, Judith Carter, Laura Cunningham, Melissa Farrelly, Andrew Freeman, Bree Galbraith, Travis Gotch, Steve Green, Greg Kammermann, Nicki Munro, Karl Newport, Katie Oxenham, Pete Paisley, Cara Reece, Karen Rusten, Justine Smith, Jenny Stott, Clint Taylor and Jeff Turpin. Thanks to Adam Bester for persevering with suggestions for improving the track monitoring transect

methodology and for coordinating the numbat release. Melissa Farrelly and Clint Taylor coordinated the feral animal control during the Wild West Releases. Reintroductions were conducted under ethics approval from the South Australian Wildlife Ethics Committee, approval numbers 42/2005, 6/2005, 19/2000, 22/99, 18/2000, 27/98, 4/99 and 2/2000.

## 2.6 REFERENCES

Australian Bureau of Meteorology 2011. Australian Bureau of Meteorology Website. <http://www.bom.gov.au>

Anthony, L.L., and D.T. Blumstein, T.D., 2000. Integrating behaviour into wildlife conservation: the multiple ways that behaviour can reduce  $N_e$ . *Biological Conservation* 95, 303–315.

Backhouse, G.N., Clark, T.W., Reading, R.P., 1994. Reintroductions for recovery of the eastern barred bandicoot *Perameles gunnii* in Victoria, Australia. in: Serena, M. (Ed.) *Reintroduction Biology of Australian and New Zealand Fauna*. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 209-218.

Bajomi, B., 2010. Reintroduction of endangered animal species: complementing the IUCN guidelines. Review protocol. <http://www.environmentalevidence.org/Documents/Draftprotocol86.pdf>. (accessed the 16th February 2011). Centre for Environmental Science, Eötvös Loránd University of Sciences, Budapest, Hungary, Pp. 10.

Beck, B. B., Rapaport L. G., Stanley-Price, M. R., Wilson, A. C., 1994. Reintroduction of captive-born animals. in: Olney, P. J. S., Mace, G. M. & Feister, A. T. C. (Eds.). *Creative conservation: interactive management of wild and captive animals*. Chapman & Hall, London, pp. 265-286.

Bester, A.J., Rusten, K., 2009. Trial translocation of the numbat (*Myrmecobius fasciatus*). *Aust. Mam.* 31, 9-16.

Bowen, Z.E., Read, J.L., 1998. Population and demographic patterns of rabbits (*Oryctolagus cuniculus*) at Roxby Downs in arid South Australia and the influence of rabbit haemorrhagic disease. *Wildl. Res.* 25, 655–62.

Brandle, R., Moseby, K. E., 1999. Comparative ecology of two populations of *Pseudomys australis* in northern South Australia. *Wildl. Res.* 26, 541-564.

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

Dickman, C.R., Mahon, P.S., Masters, P., Gibson, D.F., 1999. Long-term dynamics of rodent populations in arid Australia: the influence of rainfall. *Wildl. Res.* 26, 389-403.

Dodd, C.K., Seigel, R.A., 1991. Relocation, repatriation and translocation of amphibians and reptiles: are they strategies that work? *Herpetologica* 47, 336-350.



- Engeman, R.M., Allen, L., 2000. Overview of a passive tracking index for monitoring wild canids and associated species. *Int. Pest Mgmt. Rev.* 5, 197-203.
- Finlayson, G.R., Viera, E.M., Priddel, D., Wheeler, R., Bentley, J., Dickman, C.R., 2008. Multi-scale patterns of habitat use by re-introduced mammals: A case study using medium-sized marsupials. *Biol. Cons.* 141, 320-331.
- Finlayson, H.H., 1941. On central Australian mammals. Part III. The Muridae. *Trans. Roy. Soc. S.A.* 65, 215-231.
- Finlayson, H.H., 1961. On central Australian mammals, Part IV. The distribution and status of central Australian species. *Recs. S.A. Mus.* 14, 141-91.
- Fischer, J., Lindenmayer, D.B., 2000. An assessment of the published results of animal relocations. *Biol. Cons.* 96, 1-11.
- Friedel, M.H., 1985. The population structure and density of central Australian trees and shrubs, and relationships to range condition, rabbit abundance and soil. *Aust. Rangel. J.* 7, 130-139.
- Friend, J.A., 2008. Numbat (*Myrmecobius fasciatus*). in: Van Dyck S., Strahan, R. (Eds.) *The Mammals of Australia*. Reed New Holland, Australia, pp. 162-165.
- Gibson, D.F., Johnson, K.A., Langford, D.G., Cole, J.R., Clarke, D.E., Willowra Community, 1994. The Rufous Hare-wallaby *Lagorchestes hirsutus*: a history of experimental reintroduction in the Tanami Desert, Northern Territory. in: Serena, M. (Ed.), *Reintroduction Biology of Australian and New Zealand Fauna*. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 171-176.
- Griffith, B., Scott, J.M., Carpenter, J.W., Reed, C., 1989. Translocation as a species conservation tool: status and strategy. *Science* 245, 477-480.
- IUCN, 1998. Guidelines for reintroductions. Prepared by IUCN / SSC Reintroduction Specialist Group. IUCN, Gland, Switzerland.
- Lange, R.T., Graham, C.R., 1983. Rabbits and the failure of regeneration in Australian arid zone *Acacia*. *Aust. J. Ecol.* 8, 377-381.
- Masters, P., 1993. The effects of fire-driven succession and rainfall on small mammals in spinifex grassland at Uluru National Park, Northern Territory. *Wildl. Res.* 20, 803-813.
- Morton, S.R., 1990. The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proc. Ecol. Soc. Aust.* 16, 201-213.
- Moseby K.E., Bice, J., 2004. A trial reintroduction of the Greater Stick-nest Rat (*Leporillus conditor*) in arid South Australia. *Ecol. Mgmt. Rest.* 5, 118-124.
- Moseby, K.E., O'Donnell, E., 2003. Reintroduction of the greater bilby, *Macrotis lagotis* (Reid) (Marsupialia : Thylacomyidae), to northern South Australia: survival, ecology and notes on reintroduction protocols. *Wildl. Res.* 30, 15-27.
- Moseby, K.E., Read, J.L., 2006. The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biol. Cons.* 127, 429-437.

- Moseby, K.E., Owens, H., Brandle, R., Bice, J.K., Gates, J., 2006. Variation in population dynamics and movement patterns between two geographically isolated populations of the dusky hopping mouse (*Notomys fuscus*). *Wildl. Res.* 33, 222-232.
- Newsome, A.E., Corbett, L.K., 1975. Outbreaks of rodents in semi-arid and arid Australia: causes, preventions, and evolutionary considerations. in: Prakash, I., Ghosh, P.K. (Eds.), *Rodents and desert environments*. Junk, The Hague, Netherlands, pp. 117-153.
- Newsome, A.E., Parer, I., Catling, P.C., 1989. Prolonged prey suppression by carnivores - predator-removal experiments. *Oecologia* 78, 458-67.
- Norrdahl, K., Korpimäki, E., 1998. Does mobility or sex of voles affect risk of predation by mammalian predators? *Ecology* 79, 226-232.
- Owens, H.M., Read, J.L., 1999. Mammals of the Lake Eyre South region. Vol. 1, Part 2. in: Slater, W.J.H. (Ed.), *Lake Eyre South Monograph Series*, RGSSA, Adelaide.
- Priddel, D., Wheeler, R., 1997. Efficacy of fox control in reducing the mortality of released captive-reared Malleefowl, *Leipoa ocellata*. *Wildlife Research* 24, 469-482.
- Priddel, D., Wheeler, R., 2002. An experimental translocation of Brush-tailed Bettongs *Bettongia penicillata* to western New South Wales. Department of Conservation and Environment, Hurstville, NSW, Australia.
- Read, J.L., 1995. Recruitment characteristics of the White Cypress Pine (*Callitris glaucophylla*) in arid South Australia. *Rangel. J.* 17, 228-240.
- Read, J.L., 2004. Catastrophic drought-induced die-off of perennial chenopod shrubs in arid Australia following intensive cattle browsing. *J. Arid Environ* 58, 535-544.
- Read, J.L., Cunningham, R., 2010. Relative impacts of cattle grazing and feral animals on an Australian arid zone reptile and small mammal assemblage. *Aust. Ecol.* 35, 314-324.
- Read, J.L., Eldridge, S., 2010. An optimised rapid detection technique for simultaneously monitoring activity of rabbits, cats, foxes and dingoes in the rangelands. *Rangel. J.* 32, 389-394.
- Read, J.L., Johnston, G.R., Morley, T.P., 2011a. Snake predation thwarts trial reintroduction of threatened woma pythons, *Aspidites ramsayi*. *Oryx* in press.
- Read, J.L., Moseby, K.E., Briffa, J., Kilpatrick, A.D., Freeman, A., 2011b. Eradication of rabbits from landscape scale exclosures: pipedream or possibility? *Ecol. Mgmt. Rest.* 12, 46-53.
- Read, J., Ward, M.J., 2010. Warru Recovery Plan – Recovery of *Petrogale lateralis* MacDonnell Ranges Race in South Australia. Warru Recovery Team, South Australia, DENR, Adelaide.
- Richards, J.D., Short, J., 2003. Reintroduction and establishment of the western barred bandicoot *Perameles bougainville* (Marsupialia: Peramelidae) at Shark Bay, Western Australia. *Biol. Cons.* 109, 181-195.
- Robley, A.J., Short, J., Bradley, S., 2002. Do European rabbits (*Oryctolagus cuniculus*) influence the population ecology of the Burrowing Bettong (*Bettongia lesueur*)? *Wildl. Res.* 29, 423-429.

- Sarrazin, F., Barbault, R., 1996. Reintroduction: challenges and lessons for basic ecology. *Trends Ecol. & Evol.* 11, 474-478.
- Seddon, P. J., Armstrong, D. P., Maloney R. F., 2007. Developing the science of reintroduction biology. *Cons. Biol.* 21, 303-312.
- Seddon, P. J., 1999. Persistence without intervention: assessing success in wildlife reintroductions. *Trends Ecol. & Evol.* 14, 503.
- Short, J., 2009. The characteristics and success of vertebrate translocations within Australia. Australian Government Department of Agriculture, Fisheries and Forestry, Canberra.
- Short, J., Turner, B., 2000. Reintroduction of the burrowing bettong *Bettongia lesueur* (Marsupialia: Potoroidae) to mainland Australia. *Biol. Cons.* 96, 185-196.
- Short, J., Bradshaw, S.D., Giles, J., Prince, R.I.T., Wilson, G.R., 1992. Reintroduction of macropods (Marsupialia: Macropodoidea) in Australia: A review. *Biol. Cons.* 62, 189-204.
- Sinclair, A.R.E., Pech, R.P., Dickman, C.R., Hik, D., Mahon, P., Newsome, A.E., 1998. Predicting effects of predation on conservation of endangered prey. *Cons. Biol.* 12, 564–575.
- Soorae, P.S., 2010. Global Reintroduction Perspectives: 2010. Additional case-studies from around the globe. IUCN/SSC Reintroduction Specialist Group, Abu Dhabi, UAE. pp.
- Southgate, R., 1994. Why reintroduce the Bilby? in: Serena, M. (Ed.), *Reintroduction Biology of Australian and New Zealand fauna*. Surrey Beatty & Sons, Chipping Norton, pp. 165-170.
- Southgate, R., Possingham, H., 1995. Modelling the reintroduction of the greater bilby *Macrotis lagotis* using the metapopulation model Analysis of the Likelihood of Extinction. *Biol. Cons.* 73, 151-160.
- Spencer, P.B.S., 1991. Evidence of predation by a feral cat, *Felis catus* (Carnivora: Felidae) on an isolated rock-wallaby colony in tropical Queensland. *Aust. Mam.* 14, 143-144.
- Steen, H., 1994. Low survival of long distance dispersers of the root vole (*Microtus oeconomus*). *Annales Zoologici Fennici* 31, 271–274.
- Sutherland, W.J., Armstrong, D., Butchart, S.H.M., Earnhardt, J.M., Ewen, J. Jamieson, I., Jones, C.G., Lee, R., Newbery, P., Nichols, J.D., Parker, K.A., Sarrazin, F., Seddon, P.J., Shah, N. and Tatayah, V., 2010. Standards for documenting and monitoring bird reintroduction projects. *Cons. Letters* 3, 229-235.
- Vale, T., Schmitz, A., Fogarty, P., Bentley, J., 2004. Translocation Proposal for the reintroduction of the Burrowing Bettong or Boodie (*Bettongia lesueur*) to Yookamurra Sanctuary, Murraylands South Australia. Australia.
- Watson, A.J., Halley, M., 1999. Recovery Plan for the Eastern Barred Bandicoot *Perameles gunnii* (mainland subspecies). Department of Natural Resources and Environment, Victoria, Australia.
- Wilson, A.D., 1990. The effect of grazing on Australian ecosystems. in: Saunders, D.A., Hopkins, A.J.M. & How, R.A. (Eds.), *Australian Ecosystems: 200 years of utilization, degradation and reconstruction*. Proceedings of the Ecological Society of Australia, Surrey Beatty and Sons, Chipping Norton, Australia, pp. 235-244.

Wolf, C. M., Garland, T. J., Griffith, B., 1998. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. *Biol. Cons.* 86, 243-255.

### 3. CHAPTER THREE : THE USE OF POISON BAITS TO CONTROL FERAL CATS AND RED FOXES IN ARID SOUTH AUSTRALIA. I. AERIAL BAITING TRIALS.

Published as a research article in Wildlife Research.

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

Chapter 3- Moseby, K.E. and Hill, B.M (2011). The use of poison baits to control feral cats and red foxes in arid South Australia 1. Aerial Baiting Trials. Wildlife Research, **38**, 338-349.

**Author Contribution**

Conceived and designed the experiments- KEM

Performed the experiments including field work-KEM, BMH

Analysed the data-BMH

Contributed reagents/materials/analysis tools

Wrote the paper-KEM

**Author Declaration**

I approve the above author contribution statement and give permission for the paper/s co-authored by me to be included in the thesis by Katherine Moseby

---

Name: Brydie May Hill

Signature:

Date:

---

## Abstract

**Context.** Feral cats and foxes pose a significant threat to native wildlife in the Australian arid zone and their broadscale control is required for the protection of threatened species.

**Aims.** The aim of this research was to trial aerial poison baiting as a means of controlling feral cats and foxes in northern South Australia.

**Methods.** Eradicat™ baits or dried meat baits containing 1080 poison were distributed by air over areas of 650 to 1800 km<sup>2</sup> in trials from 2002 and 2006. Different baiting density, frequency, bait type and area were trialled to determine the optimum baiting strategy. Baiting success was determined through mortality of radio-collared animals and differences in the track activity of cats and foxes in baited and unbaited areas.

**Key Results.** Quarterly aerial baiting at a density of 10 baits per km<sup>2</sup> successfully controlled foxes over a 12 month period, whilst annual baiting led to reinvasion within four months. Despite the majority of radio-collared cats dying after baiting, a significant decline in cat activity was only recorded during one of the eight baiting events. This event coincided with extremely dry conditions and low rabbit abundance. Rabbit activity increased significantly in baited areas over the study period in comparison to control areas.

**Conclusions.** Despite trialling different baiting density, frequency and area over a five year period, a successful long-term baiting strategy for feral cats could not be developed using Eradicat™ baits or dried meat baits.

**Implications.** Broadscale control of feral cats in the arid zone remains a significant challenge and may require a combination of control methods with flexible delivery times dependent on local conditions. However, it is doubtful that current methods, even used in combination, will enable cat numbers to be reduced to levels where successful reintroductions of many threatened wildlife species can occur.

## 3.1 INTRODUCTION

The introduced feral cat (*Felis catus*) and red fox (*Vulpes vulpes*) pose a significant threat to native wildlife in Australia and are responsible for the failure of many threatened species reintroduction programs (Short *et al.* 1992; Christensen and Burrows 1994; Gibson *et al.* 1994; Southgate and Possingham 1995; Priddel and Wheeler 1997; Priddel and Wheeler 2002). The introduced red fox is also considered a key threat to many threatened species in North America (Lewis *et al.* 1999) and feral cats have been implicated in the extinction of several species in Mexico (Wood *et al.* 2002). In Australia, feral cats are thought to cause the decline and extinction of many native animals on islands (Dickman 1996) and Priddel and Wheeler (2002) found cat predation responsible for the failure of brush-tailed bettongs (*Bettongia penicillata*) to re-establish at Yathong Nature Reserve in western NSW. Control of these introduced predators is often the most important management action required for successful re-establishment of threatened mammal species in Australia (Kinnear *et al.* 2002).

A variety of methods have been used to control foxes including poison baiting (Thomson and Algar 2000), trapping and shooting. Most studies have found poison baiting to be highly effective both in Australia (Kinnear *et al.* 1998; Thomson and Algar 2000; Olsson *et al.* 2005) and overseas (Hegglin *et al.* 2004). Significant native fauna responses to fox baiting have

been recorded including increases in abundance of threatened species (Kinnear *et al.* 2002) and other native fauna such as goannas (*Varanus* spp; Olsson *et al.* 2005). Poison baiting has a long history in Australia, with most practitioners now using the poison 1080 (sodium monofluoroacetate), a derivative of the naturally occurring fluoroacetate compound found in many *Gastrolobium* and *Oxylobium* plants in Western Australia (Eason 2002). The 1080 compound is odourless, tasteless and colourless and many native species have evolved a tolerance to it (King 1990). The poison is injected into a bait substrate which is normally meat-based when used for introduced predator control.

Unfortunately feral cat control through poison baiting has often been less effective due to poor bait uptake (Risbey *et al.* 1997; Burrows *et al.* 2003; Algar and Burrows 2004; Hegglin *et al.* 2004; Olsson *et al.* 2005; Algar *et al.* 2007; Moseby *et al.* 2009a) and a cost effective, large scale control mechanism for feral cats is currently not available (Denny and Dickman 2010). Poison baiting for feral cats has been most successful in confined areas such as islands (Twyford *et al.* 2000) or in areas where alternative live prey are in low abundance (Algar *et al.* 2007). Recent research into bait attractiveness has led to the development of a soft sausage bait, designed to be more attractive to feral cats (Algar and Burrows 2004). This Eradicat™ bait has been successfully used to control cats in some areas of Western Australia however bait uptake has been found to be extremely variable with a bait density of 10 baits per km<sup>2</sup> resulting in more than 75% reduction in cat activity in some years and only 25% reduction in others (Algar and Burrows 2004; Algar *et al.* 2007). Even at higher baiting densities of 50 baits per km<sup>2</sup>, poor results have been recorded when prey species such as rabbits (*Oryctolagus cuniculus*) are in high abundance (Algar and Burrows 2004). Although the Eradicat™ bait was developed to target feral cats, it is also highly effective against foxes (Algar and Burrows 2004).

The success of poison baiting may depend on factors such as bait palatability, timing, density, delivery, frequency and baiting area. If baited areas are too small, rapid reinvasion by animals living in peripheral non-baited areas may occur. Within arid Australia, the home-range and movements of feral cats and foxes are significantly larger than in other environments (Edwards *et al.* 2001; Burrows *et al.* 2003; Moseby *et al.* 2009a), perhaps reflecting the lower productivity. Movements of more than 26 km and 45 km in three and two days respectively have been recorded suggesting that reinvasion into baited areas may be rapid and large baited areas are required (Moseby *et al.* 2009a). Thomson *et al.* (2000) found that reinvasion of a baited area by foxes occurred faster in autumn, possibly due to dispersal of juveniles. Algar and Burrows (2004) suggested that bait uptake by cats in the arid zone is higher under cool, dry conditions in late autumn and winter. Rabbit densities in the region are typically lowest during this time (Bowen and Read 1998), and previous studies have shown that highest bait uptake is during periods of low rabbit abundance (Short *et al.* 1997; Algar *et al.* 2007).

Arid Recovery is a conservation reserve in northern South Australia where rabbits, feral cats and foxes have been eradicated and excluded from a large fenced enclosure for the protection of native species (Moseby *et al.* 2009b). Radiotracking studies have shown that feral cats outside the Reserve are wild and do not rely on human contact (Moseby *et al.* 2009a), the nearest town being more than 25 km from the Reserve. The aim of this study was to determine if a cost-effective baiting regime outside the Arid Recovery Reserve could reliably and significantly reduce the activity of feral cats and foxes outside the Reserve. Reduced cat and fox activity was considered desirable in order to minimise the likelihood of foxes and feral cats breaching the fence and to increase the area of habitat available for threatened species. This study outlines the success of eight aerial poison baiting events for the feral cat and red fox over the five years, 2002-2006.



## 3.2 METHODS

### 3.2.1 STUDY AREA

The study was conducted between October 2001 and December 2006 inclusive within a 20 km radius of the Arid Recovery Reserve in northern South Australia (30°29'S, 136°53'E). The climate is hot and dry with a long term average rainfall of 166 mm per annum. The mean annual summer maximum temperature exceeds 35° C, and the mean annual winter minimum is 4° C (Olympic Dam Operations 1994).

The study area supports a variety of habitats including dunes (*Acacia ligulata* and *Dodonaea viscosa*), sandplains (*A. aneura* and *Callitris glaucophylla*), chenopod swales (*Atriplex vesicaria* and *Maireana astrotricha*), ephemeral swamps (*Eragrostis australasica*), claypans and creeklines. Taller vegetation is present on dunes, whereas swales support more open, low vegetation (Finlayson and Moseby 2004). Feral cats, red foxes and European rabbits (*Oryctolagus cuniculus*) were present throughout the study area, which is primarily used for cattle (*Bos taurus*) grazing. Feral cat and fox densities in the study region fluctuate according to seasonal conditions but averaged approximately 0.8 and 0.6 per km<sup>2</sup> respectively over a 10 year period prior to the study (Read and Bowen 2001). Regional targeted control is limited to irregular shooting by amateur shooters and some irregular poison baiting to the north of the study area for dingo (*Canis lupus dingo*) control. Rabbit densities during the study period were estimated using spotlight counts, and averaged between 51 and 55 per km<sup>2</sup> (BHP Environmental Department, unpublished data). Prior to the introduction of Rabbit Haemorrhagic Disease in 1995, rabbit density averaged between 100 and 150 per km<sup>2</sup> (BHP Environmental Department unpublished data).

The Arid Recovery Reserve lies immediately south of the dingo fence, a man-made structure built to protect sheep from dingoes. Dingoes are present to the north of the dingo fence but are sometimes baited in a 30 km buffer zone north of the fence to minimise fence breaches. The baited and control track transects used in this study included areas to the north and south of the dingo fence. For this reason the location of transects north and south of the dingo fence was included in the analysis as a covariate to determine if the presence of dingoes influenced the baiting results.

### 3.2.2 POISON BAITING

Two bait types were used in the aerial baiting trials, Eradicat™ baits and dried kangaroo (*Macropus* spp) meat baits. Eradicat™ baits were developed by the Western Australian Department of Environment and Conservation and are a semi-dried, meat product containing additives specifically attractive to cats. Eradicat™ baits weighed 20 g net (dried to 15 g) and contained 4.5 mg of 1080 (sodium monofluoroacetate). Baits were used under an experimental license held by the Western Australian Department for Environment and Conservation and the South Australian Department of Water, Land and Biodiversity Conservation. The baits were frozen until the morning of the baiting when they were thawed on mesh racks and sprayed with a residual insecticide (Coopex®) mixed with water to reduce

insect attack. Baits were then left for 1-2 hrs on the racks to 'sweat', where oils from within the sausages start to show on the surface and the outer skin becomes firm. In 2002 and 2003, baits contained red Rhodamine B dye, a biomarker which fluoresces orange under UV light and can be used to assess bait consumption by feral animals (Fisher *et al.* 1999). In 2004, dried meat baits were used instead of Eradicat™ bait and were approximately 80-120 g pieces of kangaroo meat injected with 3 mg of 1080. Baits were dried to 50% of their weight, frozen and thawed prior to use. Dried meat baits contained a lower concentration of 1080 than Eradicat™ baits as they are produced by the South Australian Department for Environment and Natural Resources for controlling foxes which are more susceptible to 1080 poisoning than cats. However, the LD50 for foxes is 0.13 mg/kg (McIlroy and King 1990) compared with 0.28 mg/kg for cats (Eason *et al.* 1992) so both baits contain enough poison to potentially kill either species.

The density, frequency and area of baiting varied from year to year in an attempt to determine the optimum baiting strategy (Table 1). Random assignment and replication of treatments was not feasible due to the requirements that the baited area be located around the Arid Recovery Reserve to protect re-introduced species. Other reasons for lack of replication included funding constraints, the remote nature of the site and the large scale required for meaningful treatments. After discussions with the bait developer and manufacturer (Western Australian Department for Environment and Conservation) we decided to initially trial a bait density of 25 Eradicat™ baits per square kilometre. This rate was based on advice from the supplier that 25 baits per square kilometre was likely to be as effective as their trialed rate of 50 baits (Dave Algar pers. comm.). Subsequent published trials in Western Australia recorded baiting success at 10, 22, 50 and 100 baits per square kilometre depending on prey availability and seasonal conditions (Burrows *et al.* 2003; Algar and Burrows 2004). Algar and Burrows (2004) suggested that reducing bait intensity below 50 per square kilometre may not reduce baiting efficacy. Additionally, we felt that the cost of baits and potential risk to non-target species rendered 50 baits per square kilometre an unrealistic and unsustainable baiting density over our landscape scale study site. Baiting was conducted in autumn or winter when prey availability was lowest. After two years of trials at 25 baits per square kilometre, dried kangaroo baits were trialled due to results suggesting some cats would ingest buried dried meat baits. Finally, a bait density of 10 Eradicat™ baits per square kilometre was trialled at a higher baiting frequency in an attempt to combat high reinvasion rates and increase the chance of baiting during key times of low food availability.

During the first year of baiting, baits were laid in a 10 km buffer zone around the outside of the Arid Recovery Reserve fenceline (Fig. 1). In subsequent years this was increased to 20 km. Baits were individually dropped from a helicopter or fixed wing aircraft along 1 km wide flight paths. Flight paths followed linear dunes as previous research had indicated that cats and foxes prefer dune habitat (Moseby *et al.* 2009a). An automated GPS recorded the location of all baits dropped during the program and ensured that no baits were dropped outside the baiting boundary. In 2002, 1400 dried kangaroo meat fox baits were buried using ground baiting two weeks prior to the aerial baiting to reduce the amount of Eradicat™ baits taken by foxes.

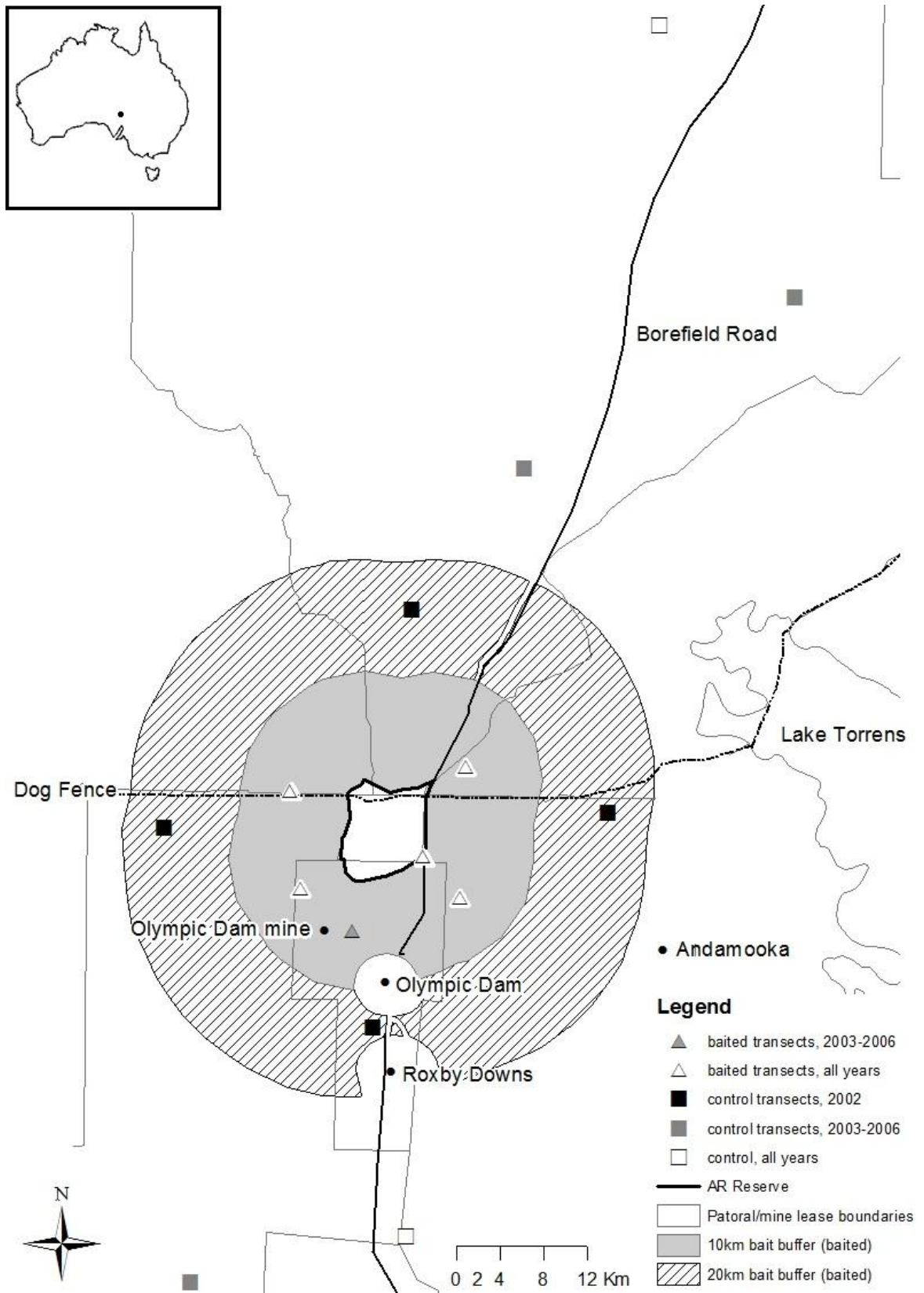
**Table 1:** Details of the aerial baiting regime used each year around the Arid Recovery Reserve, 2002-2006.

\* preceded by ground baiting 3 weeks prior, using 1400 buried dried kangaroo meat baits at an approximate density of two per km<sup>2</sup> to target foxes and thus maximise availability of Eradicat<sup>TM</sup> baits to feral cats.

Year	Bait Type	Frequency	Density (per km <sup>2</sup> )	Total baits	Area (ha)	Timing
2002	Eradicat*	annual	25	15 000	65 000	June
2003	Eradicat	annual	25	45 000	180 000	May
2004	dried meat	annual	5	9 000	180 000	May
2005	Eradicat	quarterly	10	54 000	180 000	May, Aug, Nov
2006	Eradicat	quarterly	10	36 000	180 000	Feb, May

### 3.2.3 CHANGE IN ACTIVITY

Mortality of radio-collared cats and changes in detection rates of cat and fox tracks were used to determine the success of baiting events. A series of track transects was established on vehicle tracks within the baited area and in nearby control areas. Transects established for the first baiting event in 2002 were altered for subsequent years due to an increase in the size of the baited area. In 2002, there were five transects within the baited area and six in the unbaited area (Figure 1). In subsequent years the number of transects increased to six in the baited area and decreased to five in the unbaited area (Figure 1). Transects in control areas were more than 10 km from the edge of the baited zone, baited transects were more than 5 km from the edge in 2002 and 10 km in subsequent years. Transects were a modified version of the monitoring technique established by Engeman and Allen (2000). Each transect consisted of a series of 200 m long segments on sand, separated by a distance of at least 500 m.



**Figure 1.** The location of the Arid Recovery Reserve, monitoring transects and baited buffer zones from 2002-2006. Transects symbols represent the mid-point of transects.

Segments were longer than those suggested by Engeman and Allen (2000) due to subsequent research by Read and Eldridge (2010) regarding optimum segment lengths for monitoring cats. Where possible, segments intercepted sand dune crests as previous studies have found preferential use of this habitat by cats and foxes in Australian desert environments (Mahon *et al.* 1998; Moseby *et al.* 2009a). In 2002, the number of segments in each transect varied from 10 – 20, in subsequent years 15 segments were counted in all transects (Figure 1).

Transects were driven over by four wheel drive vehicle in the late afternoon of the day preceding sampling. The tyre tread impressions obliterated older tracks and loosened the surface for increased detectability. Transects were monitored the following morning by a person walking along each segment of individual transects. The presence or absence of fresh cat, fox, rabbit, dingo and kangaroo tracks on the vehicle track were recorded for each segment. Although the number of intrusions into the plots over successive days is thought to be a more sensitive measure of population than a single binary measure (Engeman and Allen 2000), the large size of the study area and logistical constraints prevented more than one night of monitoring. For rabbits an additional record was made of the presence or absence of tracks within the first 20 metres of the right hand side tyre impressions because short sections are more sensitive to fluctuations in rabbit numbers when rabbit densities are high (Read and Eldridge 2010). If strong winds or rain occurred before transects could be checked they were resampled. All transects were checked over a one week period, occasionally this increased to two weeks when transects were affected by wind or rain. Transects were sampled a minimum of two times before and three times after each year's baiting event at intervals of 1-3 months. All transects were sampled within one month of each baiting event. Transects were sampled a total of 44 times between October 2001 and December 2006. Control transects were situated on surrounding pastoral stations and monitored using trained personnel including station managers. In most cases there was consistency of observers for each transect.

Feral cats within the baited area were captured and radio-collared prior to baiting trials in 2002, 2003 and 2006. Cats were also radio-collared in control areas more than 10 km from the baiting boundary in 2003 and 2006 and two foxes were radio-collared in the baited area in 2003. Feral cats were trapped using Victor Soft-Catch™ (No. 1.5) rubber jawed leg-hold traps (Coast to Coast Vermin Traps) or wire cage traps baited with rabbit, chicken or other meat. Two lures were used in association with the leghold and sometimes cage traps; 'pongo' (cat urine) and occasionally a Felid Attracting Phonic, 'FAP' (Westcare Electronics). Cat urine was collected from euthanased feral cats and occasionally from live domestic cats using stainless steel litter trays. During 2006, food was not used as a lure, to both minimise capture of non-target species and because food may attract hungry, inefficient hunters that were more likely to take poison baits during the subsequent baiting session.

Traps were checked early each morning, and captured feral cats and foxes were restrained using gloves and towels, and anaesthetised with a mixture of Metomidine Hydrochloride and Ketamine administered intramuscularly. The anaesthetic was reversed using Atipamezole Hydrochloride. The cats and foxes were weighed and sexed and released at point of capture. The condition of their teeth, body and reproductive organs were also noted. During 2002 and 2003, simple 50-70 g VHF radiocollars with a short whip antenna and leather belting were used (Biotelemetry Australia). In 2006, 135 g SIRTRACK (Havelock North, New Zealand) GPS data logger collars with VHF were used. The units recorded GPS fixes every four hours and were housed in epoxy resin and contained two antennas, micromouse GPS and 220 mm, 2NC gauge whip antenna. All collars were fitted with mortality sensors (40/80 ppm), triggered after 24 h without movement.

Animals were radiotracked on foot and by fixed wing aircraft, both opportunistically as well as one to four days prior to and one to three days after baiting events. If a cat or fox was found dead, the location was recorded and the general area inspected for any evidence of predation or regurgitation of baits. The animal was then relocated to a laboratory where its mouth and stomach contents were inspected for the remains of bait. In 2002, the liver and stomach of four cats suspected to have died from 1080 poisoning were sent to the Alan Fletcher Research Station in Queensland for testing. In 2003, the stomachs of four dead animals were removed and sent to Dave Algar (Western Australian Department for Environment and Conservation) who inspected them for the presence of Rhodamine B.

### *3.2.4 DATA ANALYSIS*

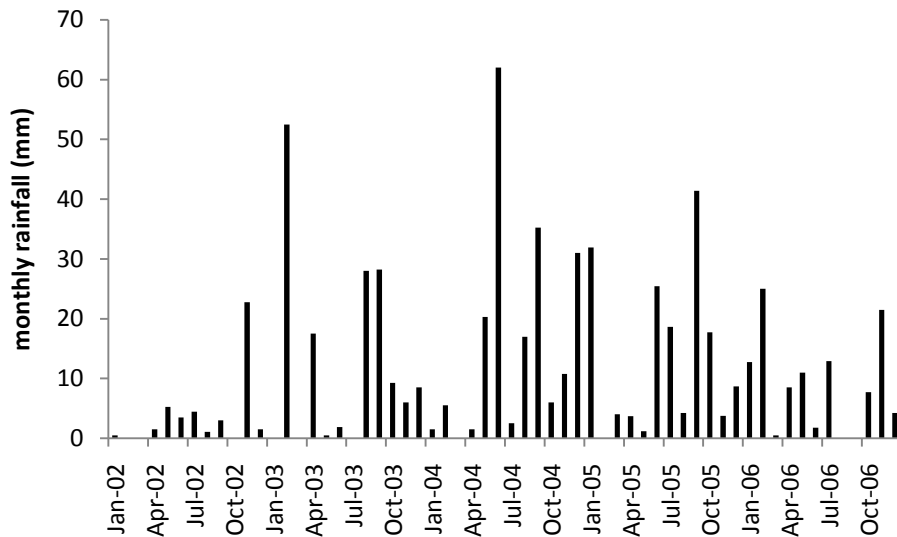
We compared transect data on control and baited transects over time to determine if baiting had a significant effect on detection rates of cats, foxes and rabbits. Detection rates were calculated at each time for each transect by dividing the number of segments with a particular species track present by the total number of segments. This proportion provides a measure of track activity for each species for each transect. These data were transformed using empirical logit transformation ( $\log((x + 0.5)/(total - (x + 0.5)))$ ), where  $x$  is the total number of segments having track activity and total is the total number of segments in the transect. Generalized linear mixed models (Galwey 2006) were used to determine significant predictor variables explaining patterns in cat, fox and rabbit detection rates. Treatment (baited or control) and time were fixed effects and site (transect) was a random effect. The significance of fixed effects was determined using Wald's Statistic (Kenward and Roger 1997). The presence of rabbit tracks in the first 20 m of the segments was used as the response variable for rabbits rather than counts from the whole transect.

Three different time scales were used for analysis. Firstly, all monitoring sessions over the entire study period were compared between baited and control transects to determine long term trends in rabbit, cat and fox detection rates and the effectiveness of baiting across all years. Secondly, the baiting events in 2002, 2003, 2004 and 2005/6 were analysed as separate baiting regimes. For each baiting regime, monthly monitoring sessions were grouped into monitoring periods and blocked into before or after baiting events (Table 2). In 2002, monitoring sessions were grouped into before, immediately after and long after the baiting event. In 2003 and 2004, monitoring sessions were grouped into before and after baiting events. For quarterly baiting in 2005 and 2006, sessions were combined and blocked into before the first baiting event, between each of the five quarterly baiting events and after the final baiting event. The interaction term between treatment and time was used to determine if baited transects responded differently to control transects after baiting. Finally, individual track monitoring sessions were compared between baited and control transects for each baiting regime. The least significant difference (lsd) term was used to identify months when significant differences occurred. This analysis allowed us to determine how long the poison baiting remained effective after each baiting event.

A factor for inside or outside the dingo fence was included as a fixed covariate in initial models to determine whether the presence or absence of dingoes influenced the effect of baiting. If the dingo fence factor was not significant it was removed from the model. Models were also run with rabbit detection as a fixed covariate to determine whether rabbit activity was influencing the effectiveness of baiting. Introduced predators are known to respond to

rabbit activity in the arid zone and rabbits are a key prey item for local foxes and cats (Read and Bowen 2001).

There was some serial dependency within sites (transects) which was accounted for by using site (transect) as a random effect. We initially explored serial dependence as a decay in an exponential way. There was some very weak evidence of exponential decay in the serial dependence component, however this did not change inferences. Predicted means derived from the models were plotted to show trends in the data. Least significant difference was used to indicate which means showed a significant difference. Bars representing least significant difference were added to graphical results.



**Figure 2.** Monthly rainfall recorded during the study.

### 3.3 RESULTS

#### 3.3.1 RAINFALL

Annual rainfall was well below average in 2002 (43 mm, Fig. 2), around the average in 2003 (152 mm), 2004 (193 mm) and 2005 (160 mm) and below average again in 2006 (105 mm).

#### 3.3.2 CATS

Despite eight baiting events, there was no significant difference in cat detection rates over time or between treatments over the five year period (Figures 3 and 4). However, a significant interaction term suggests that cat detection rates in baited and control treatments responded differently over time (Wald=8.09, df=1, P=0.004) with baited transects exhibiting considerably more variation than control transects. When baiting regimes were considered separately (2002, 2003, 2004 and 2005/6) the 2002 baiting was the only year when cat

activity declined significantly after baiting. On a monthly scale (monitoring sessions) the detection of cats in 2002 was lower on the baited transects than the control for the first and third months post baiting (Wald 25.47, df=13, P=0.02; Figure 3) but overall there was no significant interaction between treatment and time. In the first month after baiting, detection rates dropped from 17 % to 3 % on baited transects but increased from 18 % to 23 % on control transects. When the data were grouped into monitoring periods to reduce some of the variability within the monthly dataset, a significant baiting effect was observed (interaction term Wald 11.31, df =2, P<0.05; Figure 3). However, this effect only lasted for the three month monitoring period immediately post baiting.

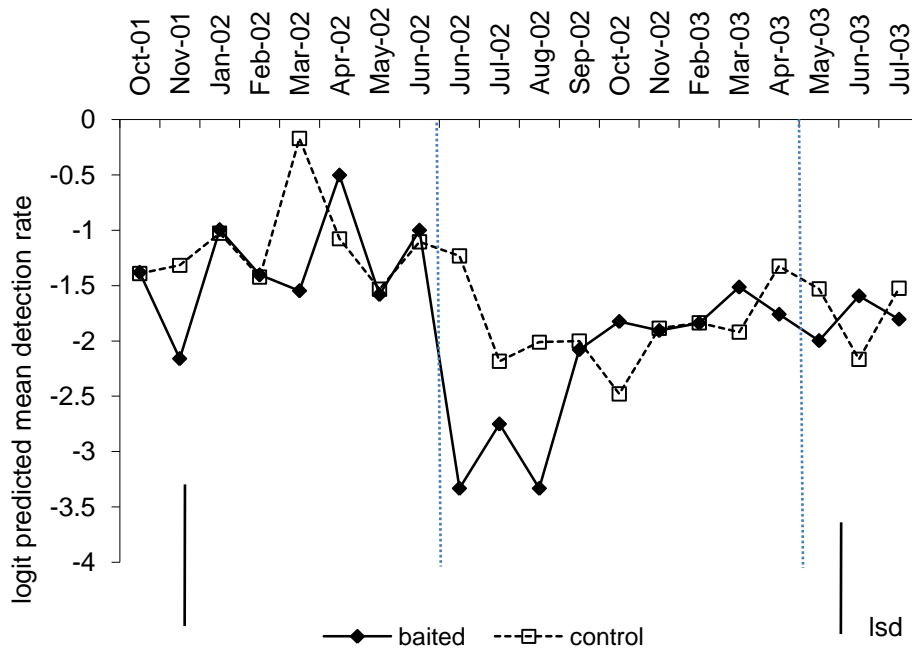
Aerial baiting was ineffective at both scales for all other baiting regimes. Interestingly, during the 2005/6 baiting regime, significantly lower detection rates were recorded in control transects during three of the post-baiting monitoring sessions. However, a non-significant interaction term indicates that this result was not related to baiting events.

Radiotracking results in 2002 accord with the decline in track detection rates recorded in baited transects with all nine radio-collared cats in the baited area dying after fox or cat baiting. Two cats were collared a month prior to the aerial baiting and just before the 1080 ground baiting for foxes, with the remaining seven collared in the three weeks between ground baiting and aerial baiting. Six cats were captured using leghold traps and three in cage traps with weights ranging from 1.5 kg to 5 kg. Two of the leghold captures were incidentally recaptured in cage traps prior to aerial baiting. Interestingly, when the cats were checked two days prior to aerial baiting, seven of the nine cats were already dead, most or all apparently killed by the buried fox baits laid two weeks earlier. The remaining two cats died within 48 hrs of the aerial baiting. One of these cats had sausage remains around its mouth and in its stomach suggesting it had died from ingesting an Eradicat™ bait. Both of the cats that died after baiting tested positive to 1080 as did one of two cats tested that died between ground baiting and aerial cat baiting.

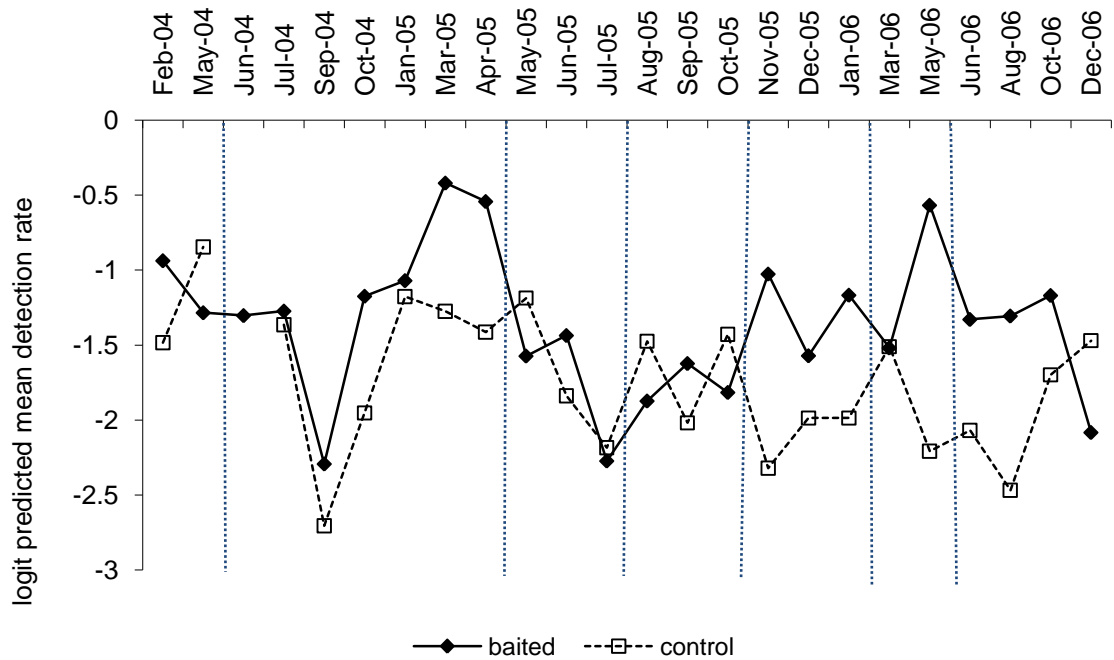
In 2003, four male and two female cats were radio-collared with two captured in leghold traps and four in cage traps. Weights ranged from 1.8 to 5.5 kg. Although poison baiting in 2003 did not lead to any significant differences between cat detection rates on baited and control transects (Figs. 3 and 4), five of the six radio-collared cats within the baited area died within three days of baiting. One radio-collared cat and one additional uncollared cat were found dead next to regurgitated baits. However, of the remaining collared cats, only one of four stomachs tested positive to the marker Rhodamine B.

In 2006, two female and four male cats were radio-collared in the baited area and six male and one female in control areas. Cats ranged in weight from 2.7 to 5.35 kg and only one cat in the baited area died in the three weeks after baiting (12 days). Two control cats died, one within two days of baiting and the other control cat moved 15 km into the baited area and died the next day, possibly after ingesting a bait. Analysis of radiocollar GPS fixes indicated that this was the only control cat that entered the baiting area. High mortality was recorded in both baited and control areas, with seven of the 11 control cats and three of the six baited cats dying between two and 60 days after baiting (Moseby *et al.* 2009a).





**Figure 3.** Logit predicted mean detection rate of cats on track transects in baited and control areas before and after baiting in 2002 and 2003. Dotted vertical lines indicate baiting events. Least significant difference (lsd) bars are shown for 2002 and 2003.



**Figure 4.** Logit predicted mean detection rate of cats on track transects in baited and control areas before and after baiting in 2004 and 2005/6. Dotted vertical lines indicate baiting events.

### 3.3.3 FOXES

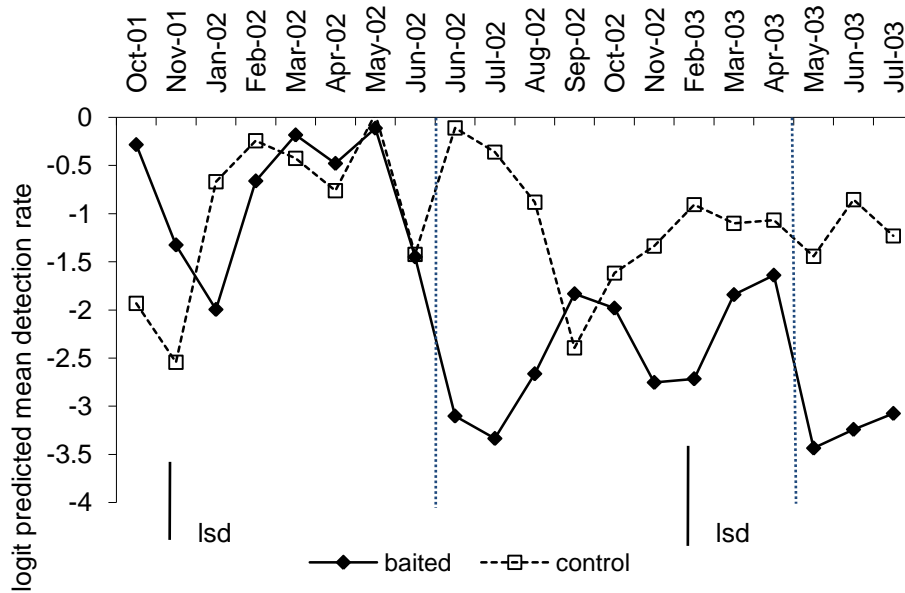
Poison baiting was highly effective in reducing fox activity in the baited area with a significant difference in fox detection rates between baited and control sites over the five year study period (Wald=12.19, df=1,  $P<0.001$ ; Figs. 5 and 6). A non-significant interaction term suggests that fox activity was consistently lower on baited transects and the overall effect of baiting was greater than the effect of reinvasion between baiting events. Towards the end of the study when quarterly baiting was implemented, detection rates on baited transects were around one third of those recorded on control transects and generally remained at less than 15% of pre-baiting rates.

When baiting regimes were analysed individually, fox detection in the baited areas consistently dropped below 10 % after all baiting events (Figs. 5 & 6). Significant interaction terms between treatment and time suggested that the decline in fox detection rates on baited transects in 2002, 2003 and 2005/6 was a response to the baiting regimes (interaction terms 2002- Wald 53.62 df 13  $P<0.001$ , 2003- Wald 11.50 df 5  $P<0.05$ , and 2005/6- Wald 31.26 df 17  $P<0.05$ ) when monthly monitoring sessions were analysed. These differences were maintained, as expected, when data were grouped and modelled at the monitoring period time scale (interaction terms 2002- Wald=32.3, df=2,  $P<0.001$ , 2003-Wald=6.64, df=1,  $P<0.01$ , 2005/6 Wald=20.45, df=5,  $P<0.001$ )

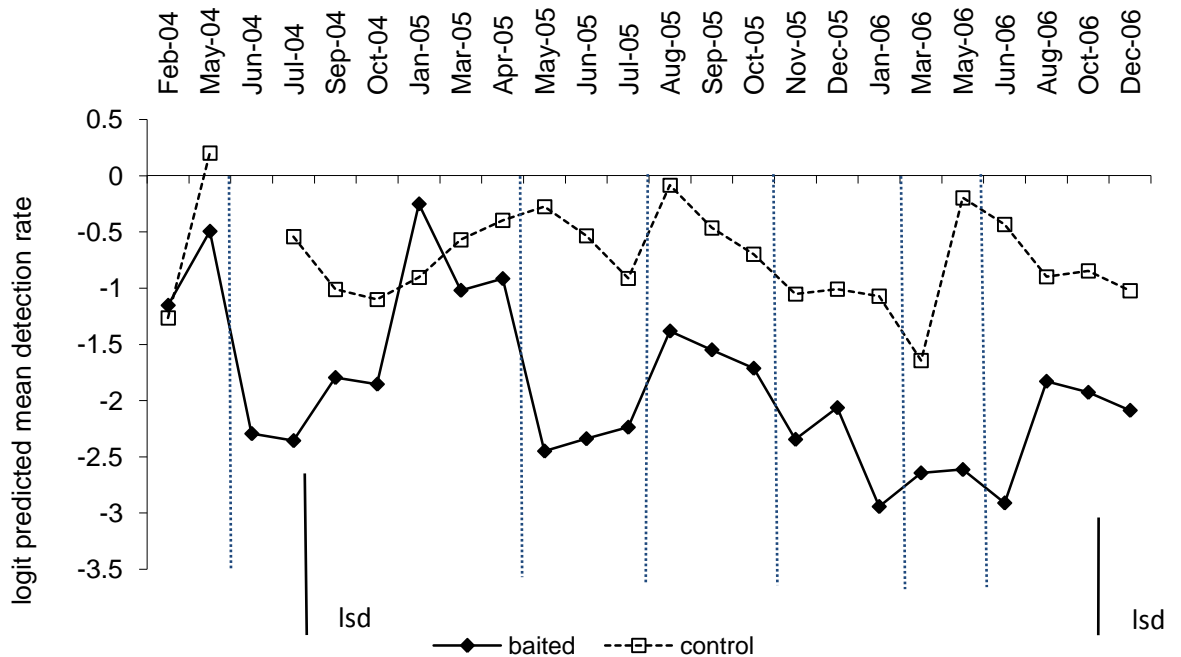
During the 2004 baiting regime, wet conditions prevented control transects from being checked in the first month after baiting. However, detection rates fell dramatically from 38 % to 9 % at baited sites after baiting and there was a significant difference in fox detection rates between treatments (Wald=6.3, df=1,  $P=0.012$ ). This difference was most pronounced in the first monitoring session after baiting (Fig. 6). The results from the model do not show a significant interaction between monitoring and treatment at a monthly or monitoring period time scale possibly due to the missing control data. A significant interaction may have been recorded if monitoring data immediately post baiting were available for control sites. Additionally, of the two foxes that were radio-collared in the baited area in 2004, one could not be located after baiting and the other died within two days of aerial baiting.

Baiting effects were short-lived with significant differences only recorded in the first few months after baiting. In 2002, fox detection rates in baited areas dropped from 47 % to 4 % but were no different to control areas at four months after baiting (Table 2, Fig 5). Baiting over a larger area in 2003 reduced fox detection rates from 16 % to 3 % and remained effective for three months. Monitoring ceased at three months after baiting and was not reinstated until nine months post-baiting by which time there was no difference between fox detection on control and baited sites. In 2004, fox detection rates at baited sites were similar to control sites at four months post-baiting (Table 2). Quarterly baiting in 2005 and 2006 produced a more sustained response but there was still some variability in fox detection on a monthly basis (Table 2).

In 2002, the only year when cat detection rates declined after baiting, there was a 91 % reduction in fox detection rates on baited transects in the month after baiting compared with an 82 % reduction for cats. Fox detection rates on baited transects were 84 % lower than control transects in the month after baiting in 2003, 82 % lower in 2004 and 81 %, 58 %, 65 % ,56 % and 87 % in 2005/6. There was no apparent relationship between bait density and magnitude of response.



**Figure 5.** Logit predicted mean detection rate of foxes on track transects in baited and control areas before and after baiting in 2002 and 2003. Dotted vertical lines indicate baiting events. Least significant difference (lsd) bars are shown for 2002 and 2003.

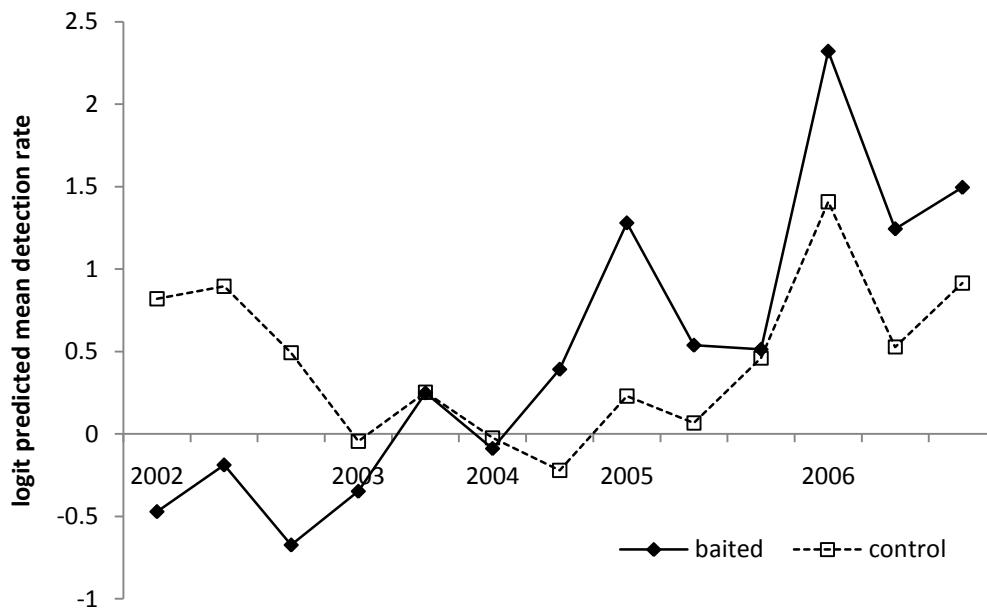


**Figure 6.** Logit predicted mean detection rate of foxes on track transects in baited and control areas before and after baiting in 2004 and 2005/6. Dotted vertical lines indicate baiting events. Least significant difference (lsd) bars are shown for 2004 and 2005/6.

### 3.3.4 RABBITS

Over the five year study there was a significant difference in rabbit detection rates over time (Wald=66.17, df=1,  $P<0.001$ ) but no significant difference between baited and control treatments. However, a significant interaction term (Wald=10.92, df=1,  $P<0.001$ ) indicates that rabbit detection in the baited and control areas responded differently over time due to rabbit detection increasing on baited transects but not control transects (Figure 7). To investigate further, years at the beginning and end of the study period were analysed separately. There were significantly more rabbits detected at control sites than at baited sites in 2002 (Wald 9.64 df=1  $P<0.05$ ). By 2005/6 there was no significant difference between treatments due to the increase in rabbits at baited sites. Results suggest that rabbits increased in the baited area over time but not in the control area.

The detection rate of rabbits was run as a covariate across the entire study period for both cats and foxes. Rabbits were significant as a single factor variable for cats (Wald =15.64, df=1,  $P<0.001$ ) suggesting that there is a relationship between detection rates of cats and rabbits. However there was no significant interaction between treatment, monitoring session and rabbits suggesting that the relationship between cats and rabbits did not influence the response of cats to baiting events. Despite this result, the only effective baiting event for cats occurred in 2002 when rabbits, their primary prey, were at their lowest for the study period. Rabbits were not significant as a covariate for foxes suggesting that baiting had a stronger influence on fox activity than rabbits.



**Figure 7.** Logit predicted mean detection rate for rabbits at baited and control sites over the 5 year study period. Monitoring sessions (months) have been blocked into monitoring periods to show overall trends.

The dingo fence was also initially run as a covariate for all of the models. Occasionally there were some minor significant effects seen between sites inside and outside of the dingo fence. However, none of these effects were related to aerial baiting and were more often mirroring the experimental design where more control sites occurred outside the dingo fence than baited sites. The dingo fence covariate was subsequently removed from the model.

**Table 2:** Track monitoring sessions used in the study including the grouped monitoring periods used for some data analysis. The dotted lines indicate baiting events and the solid lines separate different baiting regimes. ‘Between’ refers to the monthly track transects conducted between the quarterly baiting events in 2005/06. \*= a significant difference in detection rates at the P<0.05 level. Significant differences at the monitoring period time scale indicate a significant treatment by time interaction whilst significant differences at the monitoring session time scale indicate months where there was a difference in detection rates between baited and control transects.

Baiting regime	Transect monitoring period	Cats	Foxes	Transect monitoring session	Cats	Foxes
2002				Oct-01		*
2002				Nov-01		
2002				Jan-02		
2002	Before			Feb-02		
2002				Mar-02	*	
2002				Apr-02		
2002				May-02		
2002				Early Jun-02		
2002				Late Jun-02	*	*
2002	Imm After	*	*	Jul-02		*
2002				Aug-02	*	*
2002				Sep-02		
2002	After			Oct-02		
2002				Nov-02		
2003				Feb-03		*
2003	Before			Mar-03		
2003				Apr-03		
2003				May-03		*
2003	After		*	Jun-03		*
2003				Jul-03		*
2004				Feb-04		
2004	Before			May-04		

2004				Jun-04	-
2004				Jul-04	*
2004	After			Sep-04	
2004				Oct-04	
2005/06				Jan-05	
2005/06	Before			Mar-05	
2005/06				Apr-05	
2005/06				May-05	*
2005/06	Between 1	*		Jun-05	*
2005/06				Jul-05	*
2005/06				Aug-05	*
2005/06	Between 2	*		Sep-05	
2005/06				Oct-05	
2005/06				Nov-05	* *
2005/06	Between 3	*		Dec-05	
2005/06				Jan-06	*
2005/06				Mar-06	
2005/06	Between 4	*		May-06	* *
2005/06				Jun-06	*
2005/06				Aug-06	*
2005/06	After	*		Oct-06	
2005/06				Dec-06	

**Table 3:** The fate of radio-collared cats in relation to baiting events. Data from 2005/6 is taken from Moseby *et al.* 2009a. The number of cats dying after baiting refers to deaths within 14 days of baiting. Deaths after this time are more likely to be due to natural causes. No cats were radio-collared in 2004 and 2005.

\*=cats likely to have been killed by fox baits laid 3 weeks prior to aerial cat baiting.

Year	<i>Baited</i>			<i>Control</i>	
	no. cats	deaths prior	deaths after	no. cats	deaths after
2002	9	7*	2	-	-
2003	6	0	5	1	0
2006	6	0	1	7	2

### 3.4 DISCUSSION

Aerial baiting using the Eradicat™ bait was consistently effective at reducing fox but not cat activity outside the Arid Recovery Reserve. The effectiveness of cat baiting was not improved by changing the size of the baited area, baiting frequency, bait density or the seasonal timing of baiting. Statistical analysis suggested that cat activity was more strongly influenced by rabbit activity than by baiting events.

The activity of rabbits was not found to statistically influence the response of cats to baiting but this may have been at least partly related to the fact that only one baiting event occurred during a period of low rabbit activity. Cat detection rates only declined significantly after baiting in 2002, the driest year of the study and the year when rabbits were least abundant. Several other studies have found bait uptake by cats to be low, highly variable or effective only during times of low prey availability (Risbey *et al.* 1997; Short *et al.* 1997; Algar and Burrows 2004; Algar *et al.* 2007). During the 2002 baiting, most of the radio-collared cats died in the two week period between the fox baiting and aerial cat baiting and at least one death may be directly attributed to a fox bait. The high mortality suggests that cats were under severe nutritional stress and either died from starvation or were hungry enough to locate and dig up buried fox baits, a behaviour not typically recorded in other fox baiting studies (Risbey *et al.* 1997; Algar and Burrows 2004). Reduced effectiveness of baiting in subsequent trials could also be related to bait shyness by remaining cats. These cats could have received a sub-lethal dose of a decomposing bait in 2002 or naturally be more averse to consuming carrion.

Preference trials have shown that Eradicat™ baits are more palatable to cats than dried fox baits (Algar and Burrows 2004). However, although the Eradicat™ bait has been shown to be palatable at times to feral cats (Algar and Burrows 2004) it is not known if dried fox baits

would also successfully control cats if they were used during dry conditions at similarly high densities. Algar and Burrows (2004) compared fox bait and Eradocat™ uptake by cats in the Gibson Desert but the baits were used at different times and fox baits were only used at low densities. Direct comparisons of the two bait types are needed at landscape scales to determine their comparative effectiveness under different environmental conditions.

Successful control of feral cats using poison baiting requires that cats both find and ingest baits. We suggest that both of these may not have occurred effectively in our study and some other cat baiting trials. Apart from bait longevity and cat density, three main factors influence whether cats find baits in aerial baiting events, the number of baits distributed, the location of baits in the landscape and non-target uptake. A large number of baits may need to be distributed for a cat, normally an active hunter of live prey, to successfully locate an inert bait that neither moves nor emits a sound. Algar and Burrows (2004) recorded successful baiting events for cats at 22, 50 and 100 baits per square kilometre and Moseby *et al.* (2009a) used real fixes from radio-collared cats to model that 25 baits per km<sup>2</sup> square km were needed for a cat to approach one bait within three days. The lower bait density used during baiting events in 2004, 2005 and 2006 may have contributed to poor bait uptake. However, this does not explain the poor uptake in 2003 when a bait density of 25 per km<sup>2</sup> was used nor the high mortality of radio-collared cats after the 2002 fox baiting at a density of two per km<sup>2</sup>. Even at high bait densities, cats may not always effectively locate baits as they rely more on visual and audio stimuli than olfactory cues and use search images or sounds to locate prey in close proximity (K. Moseby pers obs.). This suggestion is supported by Algar *et al.* (2007) and Moseby *et al.* (2011) who recorded cat tracks travelling past baits without deviating from their line of travel. Differential habitat use by feral cats may also mean that only a portion of randomly distributed baits are functionally available to cats. Our study targeted dunes, habitat known to be preferred by cats in the region. However, cats may not use dunes randomly or may hunt in areas which are not preferred habitat. Non-target uptake was also found to be significant during ground and aerial baiting trials (Moseby *et al.* 2011) with up to 90% of baits removed by non-target species, vastly reducing baits available to cats.

Low bait ingestion rates may also have contributed to poor baiting results for feral cats. Poor bait ingestion rates may be due to cats locating baits when they are not hungry, unpalatable baits or an aversion to scavenging. Algar *et al.* (2007) and Moseby *et al.* (2011) recorded uptake rates for cats in some instances to be as low as 14% despite cats passing within 0.5 m of an Eradocat™ bait. Higher bait uptake has been recorded by researchers when using familiar foods such as birds, fish or mice (Short *et al.* 1997; Twyford *et al.* 2000; Mitchell *et al.* 2002). Catling (1988) and Paltridge *et al.* (1997) found low levels of carrion in wild arid zone cat stomachs and usually only during dry winters or droughts, suggesting a preference for live prey.

Hungry cats, stray cats or cats found at town dumps are more likely to scavenge than feral cats (Short *et al.* 2002; Risbey *et al.* 1999) and are arguably easier to trap using food as lures. The high ingestion rates of baits by radio-collared cats in 2003 despite no significant drop in track detection rates may have been partly due to the use of food-based lures and some cage traps for catching radio-collared cats. These cats may have been hungry inefficient hunters or younger inexperienced cats that were more likely to scavenge and be susceptible to baiting. Short *et al.* (2002) found cage traps caught younger cats and those that scavenged for food whilst leghold traps caught more male cats and hunters. Thomson *et al.* (2000) also found differences in bait uptake of foxes with younger foxes taking baits sooner than older foxes. Two other factors may have contributed to the discrepancy between baiting transects and radio-collared cat deaths in 2003. Firstly, some of the deaths may not have been attributable to Eradocat™ baits. Only one of the four stomachs sent to Western Australia was found to



contain the marker Rhodamine B and an insufficient number of control animals were used for death rate comparisons. The high natural mortality of both control and baited cats in 2006 indicates that high mortality is a common occurrence in the Australian arid zone (Moseby *et al.* 2009a). Similarly, the efficacy of Eradicat™ baiting in 2002 cannot be differentiated from the buried fox baits laid two weeks before as seven of the nine radio-collared cats died prior to Eradicat™ baiting. Secondly, our transects measure cat activity not cat abundance *per se*, a reduction in cat abundance could have led to an increase in the activity of remaining cats (see Christensen and Burrows 1994) or rapid reinvasion into the baited area (see Moseby *et al.* 2009a).

Fox activity was successfully reduced using both Eradicat™ poison baits and dried meat baits but quarterly baiting was required to control foxes that rapidly invaded baited areas. Foxes have been successfully controlled in many areas using 1080 poisoning (Kinnear *et al.* 2002; Algar and Burrows 2004) but reinvasion is common and varies according to season and the size of the baited area. Algar and Burrows (2004) found aerial baiting for foxes in a 160 000 ha area reduced activity for more than twelve months but other studies have found rapid reinvasion when areas of less than 10 000 ha are baited (Saunders *et al.* 1995). Kinnear *et al.* (1988) found monthly baiting was needed in areas less than 300 ha in size and Thomson *et al.* (2000) found higher reinvasion rates after six months post-baiting, particularly during autumn dispersal. Our reinvasion occurred regardless of season and faster than might have been expected from the size of the baited area. Both arid zone foxes and cats are capable of long range movements of up to 45 km in two days and have larger home ranges than their mesic counterparts (Moseby *et al.* 2009a). Arid conditions may trigger higher levels of population transience, severely hampering efforts to reduce their abundance over long time periods without regular baiting. The highest fox response was recorded in 2002 when cats also responded to baiting but there was no relationship in subsequent years between the magnitude of response and bait density or type. Fox baiting at a density of 10 baits per km<sup>2</sup> was just as effective with dried meat baits as Eradicat™ baits and much more cost-effective.

### 3.5 CONCLUSION

Despite aerial baiting trials spanning five years, eight aerial baiting events and four different baiting regimes, cat detection in the baited areas could only be significantly lowered relative to control areas for a three month period after one baiting event. Cat activity was positively related to rabbit activity and we concur with other researchers that poison baiting using Eradicat™ or dried meat baits is unlikely to be effective in areas with high rabbit abundance or when alternative prey is reliably available. Whilst successful baiting events with Eradicat™ have been documented at 50 and 100 baits per km<sup>2</sup> (Burrows *et al.* 2003), results are still variable and it is unlikely that this intensity of baiting is cost-effective or sustainable over large areas or long time periods. Fox activity could be sustainably lowered through quarterly baiting at 10 baits per km<sup>2</sup> but even when baiting a large area of 180 000 ha and reducing fox detection by up to 92 %, fox detection rates often reached 10-20 % in baited areas within just a few months of baiting. Although some threatened mammal species have responded positively to fox control (Kinnear *et al.* 2002), the presence of even low numbers of foxes can prevent the successful re-establishment of threatened species and we concur with Priddel and Wheeler (1997) that fox baiting may need to be more frequent and widespread to reduce fox populations to a level where threatened species can recover. Rabbit activity increased in the baited area relative to control areas as has been found in other cat and fox control programs (Newsome *et al.* 1989), suggesting that ongoing baiting may increase prey

availability and lead to even lower bait uptake by cats. This vicious cycle may be the nemesis of successful cat control as long as we rely on voluntary bait ingestion. The generally poor response of cats to baiting, high reinvasion rates and high densities of baits required to ensure bait uptake suggests that current baiting methods may be more suited to short-term control of cats or eradication from islands or fenced reserves. We feel that long term, broadscale cat control in areas where rabbits and other prey are present is still an aspirational target rather than an imminent outcome in the Australian arid zone.

### 3.6 ACKNOWLEDGEMENTS

This study was conducted by Arid Recovery, a conservation partnership between BHP Billiton, The University of Adelaide, the local community and the S.A. Department for Environment and Natural Resources. We would like to thank John Read for assistance with project design and for providing useful comments on the manuscript. Ross Cunningham provided patient and expert assistance with statistical analysis. Thanks must also go to David Paton and Peter Copley for their support for the project and their involvement on the Arid Recovery Steering Committee. We are indebted to the many volunteers and staff who assisted on this project including John Read, Nicki Munro, Jenny Stott, Jeff Turpin, Jason Briffa, Bree Galbraith, Michelle Thums and Adam Bester. Peter Bird provided invaluable assistance with 1080 injection and obtaining permits and Helen Crisp kindly supplied figure one. This study could not have been conducted without the dedication of the WMC Land Management section including considerable assistance from Pete Paisley, Leo and Marie McCormack and Bobby Hunter. BHP Billiton Environmental Department staff kindly assisted with data collection and provided long term rabbit datasets and comments from five referees considerably improved the manuscript.

### 3.7 REFERENCES

- Algar, D., Angus, G.J., Williams, M.R., and Mellican, A.E. (2007). Influence of bait type, weather and prey abundance on bait uptake by feral cats (*Felis catus*) on Peron Peninsula, Western Australia. *Conservation Science Western Australia* **6**, 109-149.
- Algar, D., and Burrows, N.D. (2004). Feral cat control research: *Western Shield* review- February 2003. *Conservation Science Western Australia* **5**, 131-163.
- Bowen, Z., and Read, J.L. (1998). Population and demographic patterns of rabbits (*Oryctolagus cuniculus*) at Roxby Downs in arid South Australia and the influence of rabbit haemorrhagic disease. *Wildlife Research* **25**, 655-662.
- Burrows, N.D., Algar, D., Robinson, A.D., Sinagara, J., Ward, B., and Liddelow, G. (2003). Controlling introduced predators in the Gibson Desert of Western Australia. *Journal of Arid Environments* **55**, 691-713.
- Catling, P.C. (1988). Similarities and contrasts in the diets of foxes, *Vulpes vulpes*, and cats, *Felis catus*, relative to fluctuating prey populations and drought. *Australian Wildlife Research* **15**, 307-317.

- Christensen, P. and Burrows, N. (1994). Project desert dreaming: experimental reintroduction of mammals to the Gibson Desert, Western Australia. In *'Reintroduction Biology of Australian and New Zealand Fauna'*. (Ed. M. Serena) pp. 199-207. (Surrey Beatty & Sons Pty Ltd: Chipping Norton, Australia).
- Denny, E.A. and Dickman, C.R. (2010) Review of Cat Ecology and Management Strategies in Australia. Invasive Animals Cooperative Research Centre, Canberra.
- Dickman, C.R. (1996). Overview of the Impacts of Feral Cats on Australian Native Fauna. Australian Nature Conservation Agency, Canberra.
- Eason, C.T., Morgan D.R., and Clapperton, B.K. (1992) Toxic bait and baiting strategies for feral cats. In *'Proceedings of the Fifteenth Vertebrate Pest Conference'* (Eds. J.E. Borrecco and R.E. Marsh) pp. 371-376. (University of California: Davis)
- Eason, C. (2002). Sodium monofluoroacetate (1080) risk assessment and risk communication. *Toxicology* **181**, 523-230.
- Edwards, G.P., De Preu, N.D., Shakeshaft, B.J., Crealy, I.V., and Paltridge, R.M. (2001). Home range and movements of male feral cats (*Felis catus*) in a semiarid woodland environment in central Australia. *Austral Ecology* **26**, 93-93.
- Engeman, R.M., and Allen, L. (2000) Overview of a passive tracking index for monitoring wild canids and associated species. *Integrated Pest Management Reviews* **5**, 197-203.
- Finlayson, G.R., and Moseby, K.E. (2004). Managing confined populations; the influence of density on the home range and habitat use of reintroduced burrowing bettongs (*Bettongia lesueur*). *Wildlife Research* **31**, 457-463.
- Fisher, P., Algar, R., and Sinagra, J. (1999). Use of Rhodamine B as a systemic bait marker for feral cats (*Felis catus*). *Wildlife Research* **26**, 281 – 285.
- Galwey, N. W. (2006) 'Introduction to Mixed Modelling: Beyond Regression and Analysis of Variance.' (J. Wiley and Sons: Chichester, UK).
- Gibson, D.F., Johnson, K.A., Langford, D.G., Cole, J.R., Clarke, D.E., and Willowra Community (1994). The Rufous Hare-wallaby *Lagorchestes hirsutus*: a history of experimental reintroduction in the Tanami Desert, Northern Territory. In *'Reintroduction Biology of Australian and New Zealand Fauna'*. (Ed. M. Serena.) pp. 171-176. (Surrey Beatty & Sons Pty Ltd: Chipping Norton, Australia).
- Hegglin, D., Bontadina, F., Gloor, S., Romer, J., Muller, U., Breitenmoser, U., and Deplazes, P. (2004). Baiting Red Foxes in an Urban Area: A Camera Trap Study. *Journal of Wildlife Management* **68**, 1010-1017.
- Kenward, M.G., and Roger, J.H. (1997). Small sample inference for fixed effects from restricted maximum likelihood. *Biometrics* **53**, 983-97.
- King D.R. (1990) 1080 and Australian Fauna. Western Australian Agricultural Protection Board, Perth.

- Kinnear, J.E. Onus, M.L., and Bromilow, R.N. (1988). Fox control and rock-wallaby population dynamics. *Australian Wildlife Research* **15**, 435-450.
- Kinnear, J.E., Onus, M.L., and Sumner, N.R. (1998). Fox control and rock-wallaby population dynamics - II. An update. *Wildlife Research* **25**, 81-88.
- Kinnear, J.E., Sumner, N.R., and Onus, M.L. (2002). The red fox in Australia—an exotic predator turned biocontrol agent. *Biological Conservation* **108**, 335-359.
- Lewis, J.C., Sallee, K.L., and Golightly, J.R.T. (1999). Introduction and Range Expansion of Nonnative Red Foxes. *The American Midland Naturalist* **142**, 372-377.
- Mahon, P.S., Banks, P.B., and Dickman, C.R. (1998). Population indices for wild carnivores: a critical study in sand-dune habitat, south-western Queensland. *Wildlife Research* **25**, 11-22.
- McIlroy, J.C., and King, D.R. (1990). Appropriate amounts of 1080 poison in baits to control foxes, *Vulpes vulpes*. *Australian Wildlife Research* **17**, 11-13.
- Mitchell, N., Haeffner, R., Veer, V., Fulford-Gardner, M., Clerveaux, W., Veitch, C.R., and Mitchell, G. (2002). Cat eradication and restoration of endangered iguanas (*Cyclura carinata*) on Long Cay, Caicos Bank, Turks and Caicos Islands, British West Indies. In 'Turning the Tide: The eradication of invasive species. IUCN SSC Invasive Species Specialist Group'. (Eds. C. R. Veitch and M. N. Clout.) pp. 206-212. (IUCN : Gland, Switzerland and Cambridge, UK.)
- Moseby, K.E., Stott, J., and Crisp, H. (2009a). Improving the effectiveness of poison baiting for the feral cat and European fox in northern South Australia: The influence of movement, habitat use and activity. *Wildlife Research* **36**, 1-14.
- Moseby, K.E., Hill, B.M. and Read, J.L. (2009b) Arid Recovery- A comparison of reptile and small mammal populations inside and outside a large rabbit, cat and fox-proof enclosure in arid South Australia. *Austral Ecology* **34**, 156-169
- Moseby, K.E., Read, J.L., Munro N., Galbraith, B., Newport, J., and Hill, B.M (2011). The use of poison baits to control feral cats and red foxes in arid South Australia II. Bait type, placement, lures and non-target uptake. *Wildlife Research*
- Newsome A.E., Parer I., and Catling P.C. (1989). Prolonged prey suppression by carnivores —predator-removal experiments. *Oecologia* **78**, 458-467.
- Olsson, M., Wapstra, E., Swan, G., Snaith, E., Clarke, R., and Madsen, T. (2005). Effects of long-term fox baiting on species composition and abundance in an Australian lizard community. *Austral Ecology* **30**, 899-905.
- Paltridge, R.M., Gibson, D., and Edwards, G. (1997). Diet of the feral cat (*Felis catus*) in Central Australia. *Wildlife Research* **24**, 67-76.
- Priddel, D., and Wheeler, R. (1997). Efficacy of fox control in reducing the mortality of released captive-reared Malleefowl, *Leipoa ocellata*. *Wildlife Research* **24**, 469-482.

- Priddel, D., and Wheeler, R. (2002). An experimental translocation of Brush-tailed Bettongs *Bettongia penicillata* to western New South Wales. Department of Conservation and Environment, Hurtsville, Australia.
- Read J.L., and Eldridge, S. (2010) An optimised rapid detection technique for simultaneously monitoring activity of rabbits, cats, foxes and dingoes in the rangelands. *The Rangeland Journal* **32**, 389-394.
- Read, J.L., and Bowen, Z. (2001). Population dynamics, diet and aspects of the biology of feral cats and foxes in arid South Australia. *Wildlife Research* **28**, 195-203.
- Risbey, D., Calver, M.C., and Short, J. (1999). The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. I. Exploring potential impact using diet analysis. *Wildlife Research* **26**, 621–630.
- Risbey, D.A., Calver, M.C., and Short, J. (1997). Control of Feral cats for nature conservaton. I. Field tests of four baiting methods. *Wildlife Research* **24**, 319-326.
- Saunders, G., Coman, B., Kinnear, J., and Braysher, M. (1995). Managing Vertebrate Pests: Foxes. Australian Government Publishing Service, Canberra.
- Short, J., Bradshaw, S.D., Giles, J., Prince, R.I.T., and Wilson, G.R. (1992). Reintroduction of macropods (Marsupialia: Macropodoidea) in Australia A review. *Biological Conservation* **62**, 189-204.
- Short, J., Turner, B., Risbey, D. A., and Carnamah, R. (1997). Control of feral cats for nature conservation II. Population reduction by poisoning. *Wildlife Research* **24**, 703–714.
- Short, J., Turner, B., and Risbey, D.A. (2002). Control of feral cats for Nature Conservation. III. Trapping. *Wildlife Research* **29**, 475-487.
- Southgate, R., and Possingham, H. (1995). Modelling the reintroduction of the greater bilby *Macrotis lagotis* using the metapopulation model Analysis of the Likelihood of Extinction. *Biological Conservation* **73**, 151-160.
- Thomson, P.C., and Algar, D. (2000). The uptake of dried meat baits by foxes and investigations of baiting rates in Western Australia. *Wildlife Research* **27**, 451-456.
- Thomson, P.C., Marlow, N.J., Rose, K., and Kok, N.E. (2000). The effectiveness of a large-scale baiting campaign and an evaluation of a buffer zone strategy for fox control. *Wildlife Research* **27**, 465-472.
- Twyford, K.L., Humphrey, P.G., Nunn, R.P., and Willoughby, L. (2000). Eradication of Feral Cats (*Felis catus*) from Gabo Island, south-east Victoria. *Ecological Management and Restoration* **1**, 42-49.
- Wood, B., Tershy, B.R., Hermosillo, M.A., Donlan, C.J., Sanchez, J.A., Keitt, B.S., Croll, D.A., Howald, G.R., and Biavaschi, N. (2002). Removing cats from islands in north-west Mexico. In 'Turning the Tide: The eradication of invasive species. IUCN SSC Invasive Species Specialist Group'. (Eds. C. R. Veitch and M. N. Clout.) pp. 374-380. (IUCN : Gland, Switzerland and Cambridge, UK).

#### 4.CHAPTER FOUR : THE USE OF POISON BAITS TO CONTROL FERAL CATS AND RED FOXES IN ARID SOUTH AUSTRALIA. II. BAIT TYPE, PLACEMENT AND NON-TARGET UPTAKE.

Published as a research article in Wildlife Research;

Moseby, K.E., Read, J.L., Galbraith, B., Munro, N., Newport, J and Hill, B.M. (2011). The use of poison baits to control feral cats and red foxes in arid South Australia II. Bait type, placement, lures and non-target uptake. *Wildlife Research* **38**, 350-358.

Chapter 4-Moseby, K.E., Read, J.L., Galbraith, S., Murch, N., Newport, J and Hill, B.M.  
(2011). The use of poison baits to control feral cats and red foxes in arid South Australia II:  
Bait type, placement, lures and non-target uptake. Wildlife Research 38, 350-358.

Author Contribution

Conceived and designed the experiments- KEM, JLR

Performed the experiments including field work- KEM, JLR, SG, BM i, NM, JA

Analysed the data- KEM

Contributed reagents/materials/analysis tools

Wrote the paper- KEM

Author Declaration

I approve the above author contribution statement and give permission for the paper/s co-authored  
by me to be included in the thesis by Katherine Moseby

Name: John L. Read

Signature:

Date: \_\_\_\_\_

Name: Bree Galbraith

Signature:

Date: \_\_\_\_\_

Name: Nicola Murch

Signature:

Date: \_\_\_\_\_

Name: Jenny Newport

Signature:

Date: \_\_\_\_\_

Name: Brydie M. Hill

Signature:

Date: \_\_\_\_\_

## Abstract

**Context.** Poison baits are often used to control both foxes and feral cats but success varies considerably.

**Aims.** This study investigated the influence of bait type, placement and lures on bait uptake by the feral cat, red fox and non-target species in order to improve baiting success and reduce non-target uptake.

**Methods.** Six short field trials were implemented during autumn and winter over a five year period in northern South Australia.

**Key Results.** Results suggest that poison baiting with Eradicat™ or dried kangaroo meat baits was inefficient for feral cats due to both low rates of bait detection and poor ingestion rates for those baits that were encountered. Cats consumed more baits on dunes than swales and uptake was higher under bushes than in open areas. The use of auditory or olfactory lures adjacent to baits did not increase ingestion rates. Foxes consumed more baits encountered than cats and exhibited no preference between Eradicat™ and kangaroo meat baits. Bait uptake by native non-target species averaged between 14 and 57 % of baits during the six trials, accounting for up to 90% of total bait uptake. Corvid species were primarily responsible for non-target uptake. Threatened mammal species investigated and nibbled baits but rarely consumed them. However, corvids and some common rodent species ingested enough poison to potentially receive a lethal dose.

**Conclusions.** Several factors likely contributed to poor bait uptake by cats including the presence of alternative prey, a preference for live prey, an aversion to scavenging or eating unfamiliar foods and a stronger reliance on visual rather than olfactory cues for locating food.

**Implications.** Further trials for control of feral cats should concentrate on increasing ingestion rates without the requirement for hunger through either involuntary ingestion via grooming or development of a highly palatable bait.

## 4.1 INTRODUCTION

Successful control of feral animals through poison baiting requires target animals to both find and ingest baits. The density of baits, non-target uptake and both fine-scale and habitat-scale bait placement can all influence whether a target animal successfully locates a poison bait. High bait uptake by non-target species can significantly reduce the number of baits available to target animals and bait density may have to be increased to compensate for non-target losses (Algar *et al.* 2007; Moseby *et al.* 2009). Alternatively, understanding the behavioural ecology of both target and non-target species may allow practitioners to optimise bait placement to reduce non-target uptake. Although aerial baiting prevents fine scale bait placement, flight lines can be manipulated to target preferred habitat or exclude habitat favoured by non-target species.

Once a bait is encountered by a target animal, successful bait ingestion is required to effect a kill and this is primarily influenced by bait palatability and the hunger of the target animal. Some species such as the red fox (*Vulpes vulpes*) are opportunistic feeders and will readily ingest a range of bait types including kangaroo, chicken, mice, egg and liver (MacDonald



1977; Short *et al.* 1997; van Polanen Petel *et al.* 2001). However, bait palatability has been found to influence ingestion rates with less palatable foods cached and highly palatable foods being more likely to be eaten or retrieved from a cache (MacDonald 1977; van Polanen Petel *et al.* 2001). Cats (*Felis catus*) are less likely to ingest poison baits than foxes (Risbey *et al.* 1997; Algar *et al.* 2007; Moseby *et al.* 2009) but have been found to consume kangaroo, chicken, fish and rabbit baits (Twyford *et al.* 2000; Algar *et al.* 2007). Specially formulated baits such as Eradecat™ can significantly reduce feral cat numbers (Algar *et al.* 2002a; Burrows *et al.* 2003) and have been used to eradicate cats from confined areas (Algar *et al.* 2002b). Unfortunately successful bait ingestion by cats has been found to be highly variable (Algar *et al.* 2007) and it is widely believed that they will only voluntarily ingest poison baits when prey densities are low and cats are hungry.

This study aimed to investigate both bait detection and bait ingestion by feral cats in arid northern South Australia. First, the fine-scale accessibility of aerially-dropped baits was investigated to determine the proportion of baits accessible to cats. Uptake rates by target and non-target species were then compared in different habitats and accessibility classes. Non-target uptake was also investigated along baiting transects and in areas where threatened species were present. Different lures and bait type were then tested to determine their influence on detection and ingestion rates. Trials were conducted during the cooler months when baiting in the arid zone has been found to be most successful (Algar *et al.* 2007). Reptiles, significant prey items for feral cats in summer (Martin *et al.* 1996; Read and Bowen 2001; Paltridge 2002) and a potentially significant non-target species, are also less active during this time. Results were used to suggest improvements to baiting strategies.

## 4.2 METHODS

### 4.2.1 STUDY AREA

Arid Recovery is a fenced reserve in northern South Australia (30°29'S, 136°53'E) where introduced rabbits, cats and foxes have been excluded. Four locally-extinct threatened species have been reintroduced; the greater stick-nest rat (*Leporillus conditor*), burrowing bettong (*Bettongia lesueur*), greater bilby (*Macrotis lagotis*) and western barred bandicoot (*Perameles bougainville*). The climate is arid with an average rainfall of only 166 mm a year. The dominant habitat types in the study area included mixed shrubland (*Acacia ligulata/Dodonaea viscosa*) on longitudinal dunes and low chenopod shrubland (*Maireana astrotricha/Atriplex vesicaria*) on clay inter-dunal swales. Perennial dune vegetation averages one to three metres in height and interdunal swale vegetation is usually less than 80 cm. Projected vegetation cover varies considerably depending on seasonal conditions but averages 20-30% on dunes and 20% on swales. Drainage is endoreic with ephemeral swamps and claypans filling after exceptional rainfall events. Rabbit densities during the study period were estimated using spotlight counts, and averaged between 51 and 55 per km<sup>2</sup> (BHP Environmental Department, unpublished data). Feral cat and fox densities in the study region fluctuate according to seasonal conditions but averaged approximately 0.8 and 0.6 per km<sup>2</sup> respectively over a 10 year period prior to the study (Read and Bowen 2001). Detection rates (percentage of independent track plots with spoor present) conducted between 2002 and 2006 averaged 18% for cats and 30% for foxes in unbaited areas and 19% and 16% in baited areas (Moseby and Hill 2011).

#### 4.2.2 POISON BAITS

Two bait types were used in the trials, Eradicat™ sausages and dried kangaroo meat baits. Eradicat™ baits were developed by the Western Australian Department of Environment and Conservation and were a semi-dried meat product containing additives specifically attractive to cats. Eradicat™ sausages weighed 20 g net (dried to 15 g) and contained 4.5 mg of 1080 (sodium monofluoroacetate), a naturally occurring compound that is lethal to cats and foxes. Baits were used under an experimental licence held by the Western Australian Department for Environment and Conservation and the South Australian Department of Water, Land and Biodiversity Conservation. The baits were frozen until the morning of use when they were laid outside on mesh racks and sprayed with a Coopex® solution (residual insecticide) to reduce insect attack. Once thawed, baits were left for 1-2 hrs on the mesh racks to ‘sweat’, where oils from within the sausages start to show on the surface and the outer skin. Dried meat baits were also used in one trial. These baits were 80-120 g pieces of kangaroo meat injected with 3 mg of 1080. Baits were dried to 50 % of their mass, frozen and defrosted prior to use.

Six bait trials were conducted between 2002 and 2007 to investigate the influence of bait placement, lures and bait type on target and non-target uptake (Table 1).

#### 4.2.3 BAIT PLACEMENT

In April 2002, 100 ‘cocktail frankfurts’ were dropped from a Cessna 172 aeroplane at a height of 150 m to investigate the proportion of aerially-dropped baits that would be accessible to feral cats. Commercially available frankfurts were used as a surrogate since they closely matched the Eradicat™ sausage baits for size and weight and this trial was conducted prior to the experimental licence being granted to Arid Recovery. Frankfurts were individually dropped in three passes over a 500 m section of representative dune that was vegetated with *A. ligulata* and *D. viscosa*. Flight speed matched that used in the broadscale aerial baiting events conducted around the Arid Recovery Reserve between 2002 and 2006 (Moseby and Hill 2011). A piece of reflective adhesive tape was placed around the centre of each frankfurt to enable observers to locate the baits that night using torches. Located baits were categorised according to whether they landed in the open, under foliage that did not extend to within 30 cm of ground level or within foliage that extended to within 30 cm of ground level. Baits in the first two categories were considered to be highly accessible to cats whilst baits in the ‘within foliage’ category were scored as high, medium or low accessibility. High accessibility was scored if the bait fell within 1 m of the edge of the foliage, medium between 1-2 m and low was more than 2 m into foliage.

In May 2002, unpoisoned Eradicat™ sausage baits were placed in both open and within foliage treatments in dune and swale habitats to test how bait accessibility and habitat affected bait uptake by target and non-target species (Table 1). We chose 3 x 1 km long sites spaced more than 3.5 km apart with 25 bait stations and 100 baits per site. Within a site, bait stations were spaced 40 m apart and each bait station comprised a cluster of four, 1 m x 1 m bait plots. Bait plots included two sand dune and two swale bait plots with one plot in each habitat type placed in an open area and one within foliage. Plots were swept to remove tracks and a bait placed in the centre of each plot. Those in the category of ‘within foliage’ (50 baits/site) were

exposed to accessibility treatments (distance in from open) based on the same ratios found in the frankfurt trial drop. Plots were checked each morning for three days and bait removal and animal tracks recorded. Strong wind or rain meant some plots were only checked for two nights. Chi-squared tests were used to determine if there were differences in bait uptake between dune open, dune within foliage, swale open and swale within foliage treatments.

Table 1: characteristics of the six baiting trials conducted during the study including bait type, native non-target uptake, rainfall and whether the baits used were injected with 1080 poison. Average annual rainfall is 166 mm.

\*=average of three sub-trials, E=Eradicat<sup>TM</sup>, M= dried meat baits

no.	Trial aim	Date	Bait type	Poison	Trial period (days)	native non-target uptake	Rainfall in 6 months prior (mm)
1	Influence of bait placement	May2002	E	no	3	57 % *	11
2	Non-target uptake- general	June2002	E	yes	20	-	41
3	Non-target uptake-general	May2003	E	yes	26	25 %	72
4	Non-target uptake- threatened spp	May2003	E	no	3	16 %	72
5	Comparison of bait type	Aug2002	E,M	yes	3	21 %	16
6	Influence of lures	July2007	E	yes	12	14 %	35

#### 4.2.4 TARGET VERSUS NON-TARGET UPTAKE

In 2002 and 2003, pilot trials were conducted to investigate the uptake of Eradicat™ baits by native and exotic species. The trials were timed to coincide with aerial baiting events in June 2002 and May 2003 in order to replicate the seasonal conditions and non-target abundance at the time of baiting. In June 2002, a single transect of 20 poisonous baits was placed along the Arid Recovery fenceline. In 2003, two transects each comprising 20 poisonous baits were situated 5 km apart, one along the Arid Recovery fenceline and the other along the Borefield Road, a gravel dirt road located 5 km south of the Arid Recovery Reserve. Baits were placed on sand dunes at least 200 m apart. At each bait location along the Arid Recovery fence transects, a single bait was randomly thrown 1-5 m from a stationary vehicle. On the Borefield Road transect, the driver would walk off the road for 40 m before turning towards the car and throwing the bait over their shoulder for a distance of approximately 5-10 m. To mark the site but avoid attracting corvids and other species to the bait, flagging tape was placed only at the point of bait projection and an arrow was drawn in the sand towards the bait. Baits were checked from the projection point each day for seven days and then every one to two days for up to 26 days. Where baits were missing, tracks were used to determine the species responsible for removing them.

A bait uptake trial was conducted also within the Arid Recovery Reserve in May 2003 to determine the response of three reintroduced threatened species, the greater bilby, burrowing bettong and greater stick-nest rat, to Eradicat™ baits. The sausages were not injected with 1080, but were otherwise prepared identically to those used in the aerial baiting trial. Eradicat™ uptake was investigated at 10 burrow/nest sites of each species. At each nest or burrow, 1 m<sup>2</sup> patches of sand were swept with the closest point being 50 cm away from each track runway, and one Eradicat™ bait placed in the centre of each plot. The number of baits at each burrow or nest site varied between three and seven, depending on the number of runways. The total number of baits at bettong, bilby and stick-nest rat burrow/nest sites was 37, 30 and 40 respectively, totaling 107 baits over 30 sites. The baits were checked each morning and tracks used to score each bait as ‘investigated’ where the animal had diverted more than 50 cm off the runway towards the bait but not consumed the bait, ‘nibbled’ where a very small amount (<10 %) of bait had been eaten, ‘consumed’ where more than 50 % or all of the bait was eaten or ‘no response’ where the bait was ignored or not found (with no tracks recorded within the one metre square patches). There were no instances where between 10 and 50 % of the bait was consumed. Baits were left for up to three nights and each bait received only one score. Once a bait had been investigated, nibbled or consumed by a threatened species it was removed from the trial to ensure independence of replicates.

#### 4.2.5 COMPARISON OF BAIT TYPE

In August 2002, uptake of buried and surface baits was compared in an area more than 40 km south of Arid Recovery within the unbaited control area for aerial baiting trials. The study area was on Roxby Downs Pastoral Station located inside the dingo fence where dingoes (*Canis lupus dingo*) are excluded for the protection of sheep. This trial was initiated after cats were thought to have died from consuming buried dried meat baits laid for foxes prior to the Eradicat™ aerial baiting trials in 2002 (Moseby and Hill 2011). One hundred bait sites were established 2 km apart on dunes along vehicle tracks. At each bait site, five plots were established 50 m apart and more than 10 m from the vehicle track. Three non-toxic ‘bait’

treatments and two controls were randomly assigned to each group of five plots. Bait treatments were Eradicat™ baits laid on the surface, fox baits laid on the surface and fox baits buried to a depth of 5-10 cm. All baits were placed in the centre of each 1 m x 1 m plot which was then raked and checked for animal tracks each morning for three days. Plots were scored as ‘visited’, where an animal had moved onto the plot but not ingested the bait, and ‘removed’, where the animal had moved onto the plot and ingested the bait. The two categories were mutually exclusive. The two control plots did not contain baits and one was a raked control and one an unraked control.

#### *4.2.6 THE USE OF LURES*

In July 2007, a trial was conducted to investigate whether bait uptake by cats could be improved by the addition of lures. Aerial baiting in the area had ceased in June 2006 and fox and cat spoor were regularly observed in the study area (see Moseby and Hill 2011). Twenty one bait stations were established outside Arid Recovery within 1 km of the Reserve. Bait stations were more than 500 m apart and set in seven groups of three consecutive stations. Bait stations were set within 2 m of unsealed access tracks and consisted of a 1 m long and 0.5 m wide corral surrounded on three sides by vegetation and dead logs to a height of 0.8 m. A toxic Eradicat™ bait was placed on the ground surface at the open front of each corral. Each bait station within a group was randomly assigned to either 1) no lure, 2) olfactory lure (tuna oil) or 3) auditory lure (Feline Attracting Phonic-FAP, Westcare Electronics). Treatments were rotated within a group every four days allowing each bait station to receive all treatments over a 12 day period. The tuna oil was placed in an open-topped jar containing sphagnum moss to hold the scent and prevent any scent being left behind when the lure was rotated to another bait station. All lures were placed at the rear of the corral. Sand within 10 m of each bait station was swept to clear pre-existing animal tracks. Both the bait stations and vehicle track was checked each morning for tracks and categorised into either; ‘visit’, referring to an animal deviating from the road to within 20 cm of a corral but no bait taken, ‘ignored’, referring to an animal travelling along the road only, and ‘removed’, when a bait was missing. Any bait that was taken was replaced due to the assumption that multiple bait uptake from a corral would be different cat or fox individuals because of the use of toxic baits. Total bait nights (number of baits x number of nights) was 252. The number of baits ignored, visited or removed was compared between treatments using a 3 x 3 contingency table. The number of baits removed, ignored and visited was also compared between species to investigate the proportion of uptake from target and non-target species.

### **4.3 RESULTS**

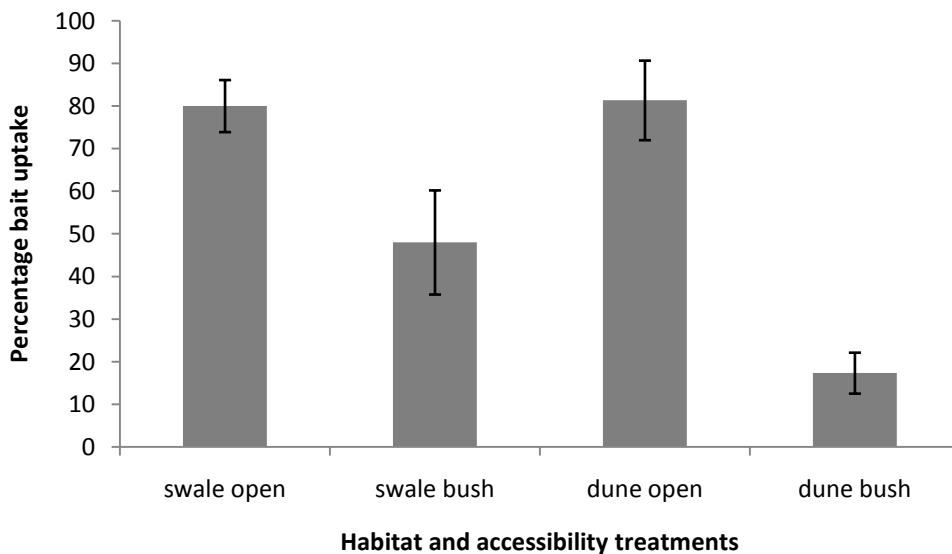
#### *4.3.1 BAIT PLACEMENT*

All 100 frankfurts dropped from the plane were retrieved within two hours of aerial deployment. Fifty one percent of baits landed in the open, 19 % under foliage greater than 30 cm from ground level and 30 % in foliage less than 30 cm from ground level. When accessibility scores were allocated to the 49 baits that fell within foliage, the majority of them

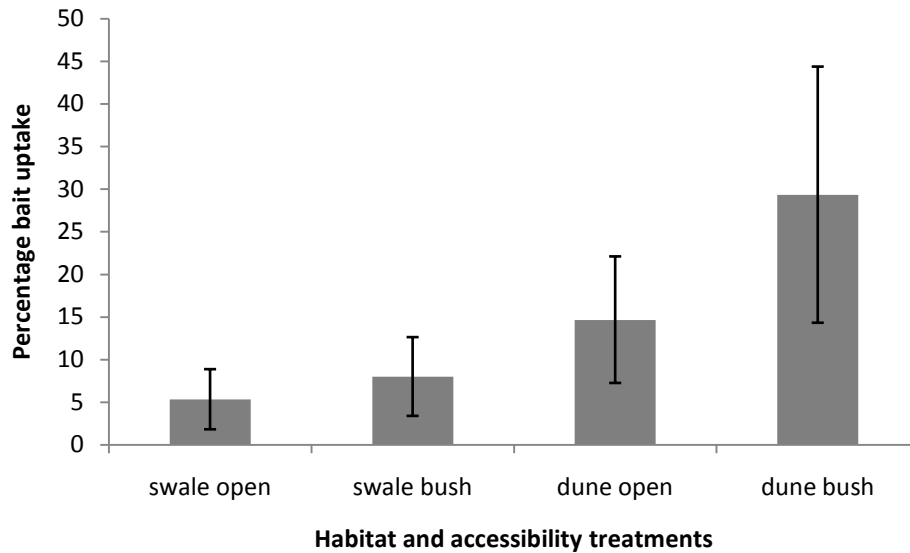
had high accessibility (75 %), 22 % had medium accessibility and only 3% of all baits were considered to have low accessibility.

When the influence of habitat type and accessibility was tested at three independent sites around the Arid Recovery Reserve there was a difference in bait uptake by both cats and corvid spp in the two habitat and two accessibility treatments (Fig. 1 and 2). Corvids removed 56 % of the total baits at all three sites with an average site bait uptake of 80 % in both the open dune and swale treatments. Differences in uptake between the four treatments were not due to chance ( $\chi^2 = 36.7$ ,  $df=3$ ,  $P < 0.001$ ) with corvids taking more baits in the open and fewer baits within foliage, particularly in dune habitat (Fig. 1). Results from all three sites were combined to increase sample size for analysis.

Bait uptake by cats was low with only 14 % of baits removed by cats during the trial. Chi-squared analysis of combined site data revealed significant differences in bait uptake by cats ( $\chi^2 = 18.1$ ,  $df=3$ ,  $P < 0.01$ ) with more baits taken from dune sites particularly within foliage (Fig. 2). In order to determine whether high corvid uptake in open sites influenced bait uptake by cats, available bait nights (number of total bait nights minus the number of baits taken by corvids) in each treatment were used to determine actual bait uptake figures for chi-squares expected values. Despite accounting for corvid uptake, cat bait uptake was still significantly different from that expected by chance ( $\chi^2 = 10.4$ ,  $df=3$ ,  $P < 0.05$ ) with more baits taken from dunes within foliage than swale and open dune sites. Wedge-tailed eagles (*Aquila audax*) took four baits and rodents one bait. An absence of sand goanna (*Varanus gouldi*) tracks at study sites suggested that they were not active during the trial. The uptake agent was unable to be determined for 20 % of baits, usually those in the within foliage treatment where tracks were harder to observe.



**Figure 1:** Average percentage bait uptake per site by corvids in dune and swale habitat in both open and within vegetation accessibility treatments. Bars indicate one standard error.

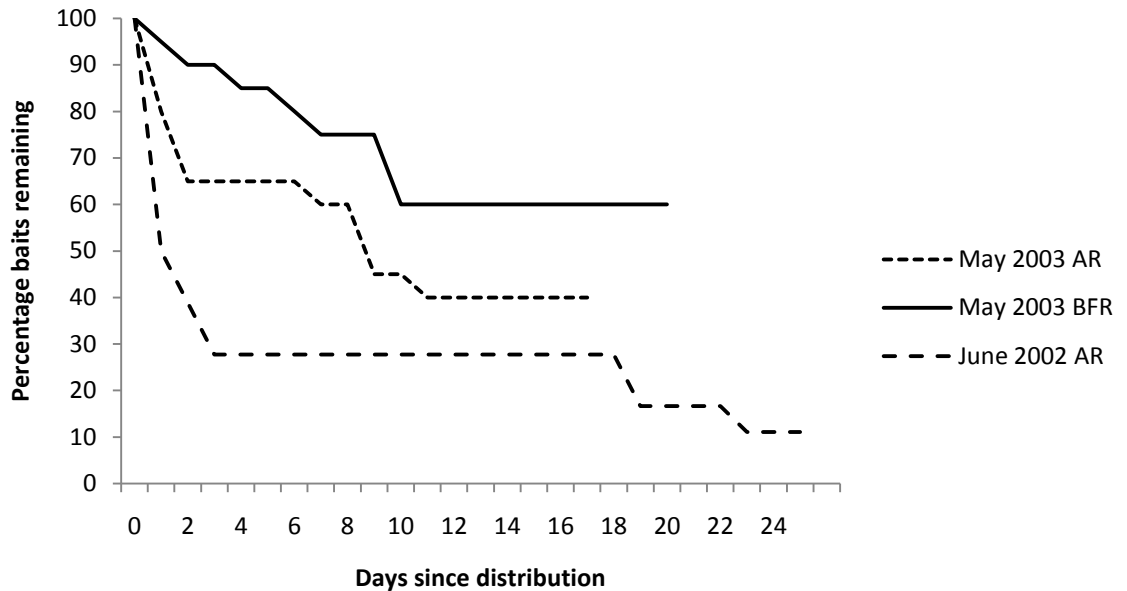


**Figure 2:** Average percentage bait uptake per site by cats in dune and swale habitat in both open and within vegetation accessibility treatments. Bars indicate one standard error.

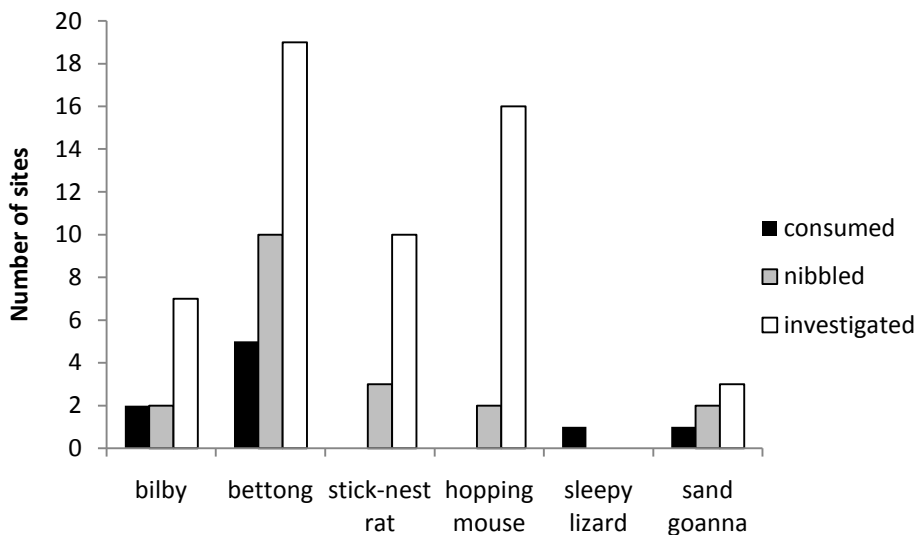
#### 4.3.2 TARGET VERSUS NON-TARGET UPTAKE

Bait uptake over time varied between the three baited transects outside the Reserve but bait uptake was highest in the first three to ten days after baiting (Fig. 3). Between 40 and 70 % of baits were taken in the first ten days with less than 20 % taken in the following seven to 16 days. The species responsible for bait uptake was not recorded in 2002 but during the two 2003 trials six of the 40 available baits (15%) were taken by corvids, one by a small cat, three by sleepy lizards (*Tiliqua rugosa*) one by a babbler-sized bird and in nine cases the uptake agent was unknown as tracks were obscured by light rain recorded during the trial.

The non-toxic Eradicat™ bait trials conducted inside the Arid Recovery Reserve compared bait uptake by re-introduced and other native species. Ninety percent of the baits were investigated by native animals but only 16 % (18 of 107 baits) were consumed (Fig. 4). Bettongs consumed 12 baits at five sites, bilbies consumed three baits at two sites, one goanna ate two baits at one site and a sleepy lizard consumed one bait (Fig. 4). Bettongs found more baits than all other species combined, and investigated, nibbled or consumed baits at 26 (87 %) of the 30 shelter sites tested. Greater bilbies investigated baits at seven sites but consumed baits at only two of their own shelter sites. Stick-nest rats investigated or nibbled baits at 13 sites including eight of their shelter sites but did not fully consume any baits.



**Figure 3:** The percentage of baits remaining each day along three baiting transects between 2002 and 2003. Twenty baits were placed along each transect. AR= Arid Recovery Reserve fenceline, BFR= Borefield Road.



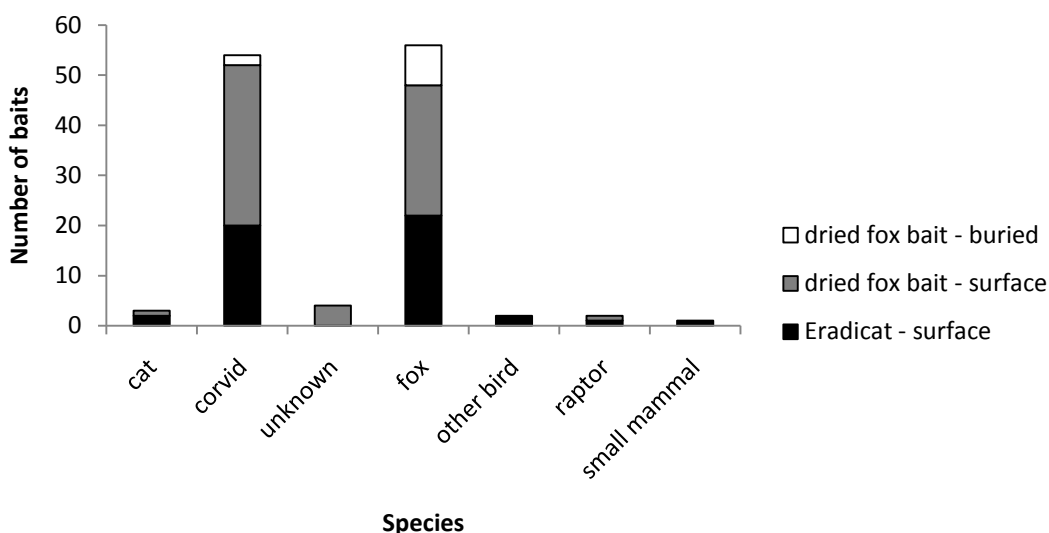
**Figure 4:** The number of sites where baits were nibbled, consumed or investigated by each species during the trial. 107 baits were placed over 30 shelter sites including 10 bilby burrows, 10 bettong burrows and 10 rat nests.

#### 4.3.3 COMPARISON OF BAIT TYPE

When bait uptake was compared between Eradicat baits and surface and buried fox baits, a total of 48 (48 %) surface Eradicat™ baits, 64 (64 %) surface fox baits and 10 (10%) buried fox baits were removed over the three day period (Table 2). Foxes and corvids removed the most baits with 19 % and 18 % of baits taken respectively (Fig. 5). Fox and corvid ingestion rates for surface fox baits and surface Eradicat™ baits were similar. Cats only removed two



Eradicat™ baits and one surface fox bait. The percentage of total baits encountered that were eaten was high in foxes (88 %) and corvids (92 %) but low in cats (25 %) and other groups such as rodents, lizards and other birds (Table 2). Higher visitation at surface bait sites (82 and 88 visits) compared with control (25 and 41 visits) and buried bait sites (18 visits) suggests that animals were attracted to the baits laid on the surface. Buried fox baits recorded the least amount of non-target uptake but also the lowest uptake by foxes and cats. Visits to buried fox baits by cats and foxes were no greater than visits to control sites without baits. Although low sample size prevented statistical analyses, visitation at raked and unraked control sites was similar suggesting that raking was unlikely to have significantly affected visitation rates.



**Figure 5:** A comparison of the number of poison baits taken by different animal groups in buried and surface bait treatments. The total number of available baits was 300.

Table 2: The number of baits that were visited and not removed, or visited and removed for each bait and control treatment. The two categories are mutually exclusive. n=100 for each treatment.

<i>Species</i>	<i>Eradicat™</i>		<i>Fox bait</i>		<i>Fox bait</i>		<i>Control</i>	<i>Control</i>
	<i>surface</i>		<i>surface</i>		<i>buried</i>		<i>raked</i>	<i>unraked</i>
	visit	removed	visit	removed	visit	removed	visit	visit
Cat	2	2	5	1	2	0	0	2
Fox	2	22	2	26	4	8	16	24
Corvid	4	20	1	32	0	2	9	15
Other/unknown	26	4	16	5	2	0	-	-

#### 4.3.4 THE USE OF LURES

When auditory and olfactory lures were used in an attempt to improve bait uptake, 39 baits were removed over 252 bait nights yielding an overall uptake of 15.5 %. Non-target species accounted for 90 % of all bait uptake with only four baits removed by foxes and two by cats. The majority of non-target uptake was due to corvids (28 baits totalling 11 % bait uptake) with four baits removed by the spinifex hopping mouse (*Notomys alexis*) and three by sleepy lizards.

Lures did not significantly increase bait uptake by cats and foxes. Cat and fox uptake data had to be combined due to the small sample size (Table 3). There was no significant difference ( $\chi^2 = 1.2$ , d.f. = 2,  $P > 0.05$ ) in the proportion of cats and foxes that visited, ignored and removed baits between the different lure types (auditory, olfactory or none). The three lure types were then combined and the proportion of baits visited, ignored and removed by cats and foxes compared with an expected equal probability in each category using chi-squared analysis. There was a significant difference in the proportion of cats and foxes in the three visit types ( $\chi^2 = 8.9$ , df=2,  $P < 0.001$ ) with more baits ignored and fewer baits removed by cats and foxes. Cats walked past nearly half of the baits that were within 5 m despite two thirds of them being associated with lures. Only 14 % (2 out of 14) of baits that were encountered by cats were ingested. Foxes also either failed to notice or were not interested in 60 % of the baits that they walked past and only consumed 50 % of the baits that they visited.

Table 3: The number of baits ignored, visited and not removed, or visited and removed by cats and foxes during the lure trial. Categories are mutually exclusive. Ignored refers to the animal moving along the adjacent vehicle track without deviating towards the bait. Total bait nights is 252.

<i>Species</i>	<i>Visit</i>	<i>Removed</i>	<i>Ignored</i>
Cat	12	2	10
Fox	4	4	12
<b>Both</b>	<b>16</b>	<b>6</b>	<b>22</b>

#### 4.4 DISCUSSION

Despite the presence of cats throughout the study area (see Moseby *et al.* 2011) and trials being conducted during the cooler autumn/winter months as suggested by Algar and Burrows (2004), bait uptake by feral cats at Roxby Downs was low with the majority of baits taken by non-target species. Cats both failed to detect most baits used in the trials as well as recording low ingestion rates for those baits that were encountered. Ingestion rates varied from as high as 50 % to as low as 14 %. Studies by Algar *et al.* (2007) and Moseby *et al.* (2009) found bait uptake rates to be highly variable (0-70 %) and often lower than 20 %. Risbey *et al.* (1997)

also found that cats approached fishmeal and digest (an additive used by pet food companies to enhance cat food palatability) baits but rarely consumed them. Several studies have recorded higher bait uptake when using familiar foods (Short *et al.* 1997; Twyford *et al.* 2000; Mitchell *et al.* 2002) and Algar *et al.* (2007) suggests that cats only eat when they are hungry so encountering baits also needs to coincide with a period of hunger. The poor baiting success recorded in many poison baiting programs for feral cats is likely to be due to a combination of failure to locate baits (even when passing within a few metres), the presence of alternative live prey (Algar and Burrows 2004) and an aversion to scavenging or consuming unfamiliar foods (see Catling 1988; Risbey *et al.* 1999; Short *et al.* 2002).

Bait ingestion rates by foxes were much higher than those recorded for cats but they still varied and ranged from 50 % to 93 %. Although not directly comparable, these figures are similar to the rate of bait uptake by fox populations that have been reported in other Australian studies (58.3% Fleming 1997; 92% Marks and Bloomfield 1999). Foxes readily consumed Eradicat™ baits as had been found by Algar and Burrows (2004).

Although limited by small sample size, the use of auditory and olfactory lures did not increase ingestion rates in our trial. Clapperton *et al.* (1994) found some olfactory lures such as catnip and matatabi attracted cats but did not determine their influence on bait uptake. Algar *et al.* (2007) found ingestion rates did not increase with auditory lures but several researchers have successfully increased bait uptake using visual lures (Friend and Algar 1995; Algar and Sinagra 1996; Algar *et al.* 2007). Cats are known to use visual and auditory stimuli more than olfactory stimuli when hunting for food (Commonwealth of Australia 2007). It is unlikely that these stimuli would be triggered by an inert, unfamiliar Eradicat™ bait, suggesting that bait presentation requires further research and ideally would closely match the appearance and behaviour of prey.

Bait uptake was highest on dunes, a result supported by local radiotracking studies that found cats prefer this habitat type (Moseby *et al.* 2009). Interestingly, bait uptake was also higher when baits were placed within vegetation rather than out in the open, a result that was partly explained by lower bait uptake by non-targets in this habitat but was also significant when the influence of non-target uptake was removed. The hunting strategies employed by feral cats may include searching within vegetation for live prey such as rodents and birds. Rodents are known to prefer to forage in areas with more cover (Parmenter and MacMahon 1983; Taraborelli *et al.* 2003) and birds and reptiles may be easier to stalk and catch within vegetation. Cats may also use cover as protection from predators such as dingoes and wedge-tailed eagles.

Bait uptake by native non-target species ranged between 14 and 57 % of baits in the six trials and often accounted for more than 90 % of the total baits consumed. Corvids were the most significant non-target species, consuming or removing large numbers of both toxic and non-toxic baits. Both Australian ravens (*Corvus coronoides*) and little crows (*corvus bennetti*) are common residents in the study region (Read *et al.* 2000). No dead birds were observed during the trials and corvids remained common in the area, suggesting that few birds received a lethal dose of 1080. However, Australian ravens and little crows would only need to ingest one and two Eradicat™ baits respectively to reach their LD50 and potentially receive a lethal dose of toxin (McIlroy 1984).

Corvid uptake was highest in the unpoisoned bait placement trial that was conducted prior to aerial baiting with comparatively lower rates in future trials. This may have been partly due to corvids learning to avoid poison baits and also partly due to the method of bait deployment in earlier trials, where closely-spaced baits and quad bikes made it easier for birds to watch and follow observers. However, corvids also located and ingested baits that were away from

human interference and even within vegetation. Corvids removed both Eradicat™ baits and unburied dried meat fox baits but rarely found buried fox baits. Algar *et al.* (2007) recorded an average non-target uptake of 22 %, with corvids and varanids the most common species. All of our trials were conducted during the cooler months and it is likely that non-target uptake would be much higher during the summer months when more reptiles, particularly varanids, are active.

The impacts of high non-target uptake include both reduction in bait availability to target species such as cats and a possible decline in abundance of 1080-sensitive non-target species. Spinifex hopping mice were found to nibble baits inside Arid Recovery and remove baits both in our lure trials and other baiting programs (Algar *et al.* 2007). In general, this species did not consume more than 10 % of the bait and spinifex hopping mice are known to be moderately tolerant to 1080 (LD50 = 32.7 mg/kg, King 1990) but the higher dose of 1080 in the sausage baits compared with standard fox baits means this species only has to ingest approximately one quarter of a bait to receive the LD50 dose of the toxin. The other abundant rodent species in the study area, *Pseudomys bolami*, is closely related to species, *P. hermansburgensis*, which needs to ingest as little as 8 % of an Eradicat™ bait to receive its LD50 dose. Sleepy lizards regularly ingested baits during our trials and other lizards such as varanids have been found to ingest Eradicat™ baits, often in high numbers (KM pers. obs.; Algar *et al.* 2007). Although sleepy lizards and goannas are extremely tolerant to 1080 (McIlroy 1985) and would have to ingest approximately 21 and 7 Eradicat™ baits respectively to receive a potentially lethal dose, these species could significantly reduce bait encounter rates for target species. Encouragingly, most threatened species used in our trials nibbled or investigated baits rather than consumed them but this may have been partly related to the good seasonal conditions stimulated by 50 mm of rain recorded 3 months prior to the trial. Burrowing bettongs ingested the most baits within Arid Recovery and are known to scavenge and consume a wide variety of food items (Robley *et al.* 2001; Bice and Moseby 2008). Some progress has been made on the development of a bait suspension device which may reduce uptake by varanids and rodents (Algar and Brazell 2008) but this is time consuming and not applicable to aerial baiting techniques.

One feral cat was found to remove a surface dried fox bait, supporting the suggestion by Moseby and Hill (2011) that cats will eat dried fox baits if they are hungry. Corvids and foxes did not show a preference for Eradicat™ or dried meat baits but consumed more baits on the surface than buried baits. Although fox bait uptake in buried bait plots was lower than surface baits, non-target uptake was only 20 % compared with 50 % for surface Eradicat™ baits and 62 % for surface fox baits. The high reduction in non-target uptake may justify burying baits in fox control programs although it is likely to also reduce uptake by feral cats.

Trials ranged in duration from three to 26 days and results from some of the shorter trials may have been improved by extending the trial period. However, the high proportion of non-target uptake and poor bait ingestion rates recorded by cats in the shorter trials suggests that uptake rates by feral cats are unlikely to have significantly improved over time.

#### 4.5 CONCLUSION

Despite the successful baiting of feral cats in some areas (Short *et al.* 1997; Algar and Burrows 2004), the high non-target uptake and risk to sensitive species coupled with the low bait detection rates and poor ingestion rates by cats suggests that Eradicat™ aerial baiting is an inefficient and ineffective broadscale control technique for feral cats in the Roxby Downs

region. Reasons for poor bait uptake by cats may include high alternative prey such as rabbits, an aversion to scavenging unless food resources are low, failure to recognize the bait as a food source, suspicion of unfamiliar foods and low reliance on olfactory senses to locate food. The main reason cited for poor bait uptake by feral cats is high numbers of alternative prey such as rabbits (e.g. Short *et al.* 1997; Algar *et al.* 2007). However, even when rabbits were in low abundance, Algar *et al.* (2007) found average bait ingestion rates by feral cats of only 28 %. Low ingestion rates suggest that significantly increasing bait density to offset high non-target uptake as recommended by Algar *et al.* (2007) may improve bait detection but is unlikely to increase bait ingestion or efficiency and could lead to high non-target impacts. The highly variable results obtained from poison baiting trials (Algar and Burrows 2004; Algar *et al.* 2007; Moseby and Hill 2011), particularly in areas where rabbits are abundant, suggests that improvements in both detection and ingestion rates are required before poison baiting becomes an effective long term control mechanism for feral cats. Detection rates could be improved through developing effective visual lures or more closely investigating the influence of bait placement. For example, placing baits under vegetation on dunes may improve detection by feral cats and reduce non-target uptake. However, improving ingestion rate is arguably easier and more important as it will minimise the number of baits required for successful control leading to higher bait efficacy, lower costs and lower non-target impacts. If bait ingestion could be assured for every incidence of bait detection then baiting could become a reliable, long term method of cat control. Further trials should concentrate on increasing ingestion rates without the requirement for hunger. This could be done either through involuntary bait ingestion via grooming (see Read 2010) or through the development of a highly palatable bait that stimulates ingestion regardless of hunger.

#### 4.6 ACKNOWLEDGEMENTS

This study was conducted by Arid Recovery, a joint conservation initiative between BHP Billiton, the community, The University of Adelaide and The S.A. Department for Environment and Natural Resources. Trials were conducted under permits 14/2002 and 9/2006 from the South Australian Wildlife Ethics Committee. Significant input was received from Arid Recovery staff, students and volunteers. In particular, the authors would like to thank Michelle Thums, Karl Newport, Renee Sternberg and Peter Paisley. Dave Algar from the W.A. Dept for Environment and Conservation kindly assisted with obtaining experimental permits and commented on experimental design. Thanks to Peter Copley and Peter Bird for assistance with South Australian permits and their support for the trials. Five referees provided useful comments on the manuscript.

## 4.7 REFERENCES

- Algar, D. and Sinagra, J.A. (1996). Methods of broadscale control of feral cats in Western Australia. Unpublished report to the Australian Nature Conservation Agency, Feral Pests Program Project No. 11. ANCA, Canberra.
- Algar, D.A., Angus, G.J., Fuller, P.J. and Onus, M.L. (2002a). Operation Wanjarri: optimising feral cat baiting strategies in the arid zone. Report to Environment Australia. Department of Conservation and Land Management, Western Australia.
- Algar, D.A., Burbidge, A.A. and Angus, G.J. (2002b). Cat eradication on Hermite Island, Montebello Islands, Western Australia. In 'Turning the Tide: Proceedings of the international conference on the eradication of island invasives'. (Eds. C. R. Veitch and M. N. Clout.) pp. 14-18. (IUCN: SSC Invasive Species Specialist Group: Gland, Switzerland and Cambridge, UK).
- Algar, D. and Burrows, N.D. (2004). Feral cat control research: *Western Shield* review-February 2003. *Conservation Science Western Australia* **5**, 131-163.
- Algar, D., Angus, G.J., Williams, M.R., and Mellican, A.E. (2007). Influence of bait type, weather and prey abundance on bait uptake by feral cats (*Felis catus*) on Peron Peninsula, Western Australia. *Conservation Science Western Australia* **6**, 109-149.
- Algar, D. and Brazell, R.I. (2008). A bait-suspension device for the control of feral cats. *Wildlife Research* **35**, 471-476.
- Bice, J.K. and Moseby, K.E. (2008). Diet of the re-introduced Greater Bilby (*Macrotis lagotis*; Peramelidae) and Burrowing Bettong (*Bettongia lesueur*; Potoroidae) in the Arid Recovery Reserve, Northern South Australia. *Australian Mammalogy* **30**, 1-12.
- Burrows, N.D., Algar, D., Robinson, A.D., Sinagra, J., Ward, B., and Liddlelow, G. (2003). Controlling introduced predators in the Gibson Desert of Western Australia. *Journal of Arid Environments* **55**, 691-713.
- Catling, P.C. (1988). Similarities and contrasts in the diets of foxes, *Vulpes vulpes*, and cats, *Felis catus*, relative to fluctuating prey populations and drought. *Australian Wildlife Research* **15**, 307-317.
- Clapperton, B.K., Eason, C.T., Weston, R.J., Woolhouse, A.D., and Morgan, D.R. (1994). Development and testing of attractants for feral cats, *Felis catus* L. *Wildlife Research* **21**, 389-399.
- Commonwealth of Australia (2007). Background document for the threat abatement plan for predation by feral cats. Department of the Environment and Water Resources, Canberra.
- Denny, E.A., and Dickman, C.R. (2010) Review of Cat Ecology and Management Strategies in Australia. Invasive Animals Cooperative Research Centre, Canberra.

- Fleming, P.J.S. (1997). Uptake of baits by red foxes: implications for rabies contingency planning. *Wildlife Research* **24**, 318-346.
- Friend, J.A., and Algar, D. (1995). Methods of broad-scale cat control and fox control at a numbat reintroduction site. Unpublished report to The Australian Nature Conservation Agency, Canberra.
- King, D.R. (1990). 1080 and Australian Fauna. Western Australian Agricultural Protection Board, Perth.
- MacDonald, D.W. (1977). On food preference in the red fox. *Mammal Review* **7**, 7-23.
- Marks, C.A., and Bloomfield, T.E. (1999). Bait uptake by foxes (*Vulpes vulpes*) in urban Melbourne: the potential of oral vaccination for rabies control. *Wildlife Research* **26**, 777-787.
- Martin, G.R., Twig, L.E., and Robinson, D.J. (1996). Comparison of the diet of feral cats from rural and pastoral Western Australia. *Wildlife Research* **34**, 475-84.
- McIlroy, J.C. (1984). The sensitivity of Australian animals to 1080 poison VII. Native and introduced birds. *Australian Wildlife Research* **11**, 373-85.
- McIlroy, J.C. (1985). The sensitivity of Australian animals to 1080 poison VIII. Amphibians and reptiles. *Australian Wildlife Research* **12**, 113-18.
- Mitchell, N., Haeffner, R., Veer, V., Fulford-Gardner, M., Clerveaux, W., Veitch, C.R. and Mitchell, G. (2002). Cat eradication and restoration of endangered iguanas (*Cyclura carinata*) on Long Cay, Caicos Bank, Turks and Caicos Islands, British West Indies. In 'Turning the Tide: Proceedings of the international conference on the eradication of island invasives'. (Eds. C. R. Veitch and M. N. Clout.) pp. 206-212. (IUCN: SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK).
- Moseby, K.E., Stott, J., and Crisp, H. (2009). Improving the effectiveness of poison baiting for the feral cat and European fox in northern South Australia: The influence of movement, habitat use and activity. *Wildlife Research* **36**, 1-14.
- Moseby, K.E. and Hill, B.M. (2011). The use of poison baits to control feral cats and red foxes in arid South Australia 1. Aerial Baiting Trials. *Wildlife Research*
- Paltridge, R. (2002). The diets of cats, foxes and dingoes in relation to prey availability in the Tanami Desert, Northern Territory. *Wildlife Research* **29**, 389-403.
- Parmenter, R. R., and MacMahon, J. A. (1983) Factors determining the abundance and distribution of rodents in a shrub-steppe ecosystems: The role of shrubs. *Oecologia* **59**, 145-56.
- Read, J.L., Ebdon F.R., and Donohoe, P. (2000). The terrestrial birds of the Roxby Downs Area: a ten year history. *South Australian Ornithologist* **33**, 71-83.
- Read, J.L., and Bowen, Z. (2001). Population dynamics, diet and aspects of the biology of feral cats and foxes in arid South Australia. *Wildlife Research* **28**, 195-203.
- Read, J.L. (2010). Can fastidiousness kill the cat? The potential for target-specific poisoning of feral cats through oral grooming. *Ecological Management and Restoration* **11**, 230-233.

- Risbey, D.A., Calver, M.C., and Short, J. (1997). Control of Feral cats for nature conservaton. I. Field tests of four baiting methods. *Wildlife Research* **24**, 319-326.
- Risbey, D.A., Calver, M.C., and Short, J. (1999). The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. I. Exploring potential impact using diet analysis. *Wildlife Research* **26**, 621–630.
- Robley, A.J., Short, J., and Bradley, J.S. (2001). Dietary overlap between the burrowing bettong (*Bettongia lesueur*) and the European rabbit (*Oryctolagus cuniculus*) in semi-arid coastal Western Australia. *Wildlife Research* **28**, 341-349.
- Short, J., Turner, B., Risbey, D. A., and Carnamah, R. (1997). Control of feral cats for nature conservation II. Population reduction by poisoning. *Wildlife Research* **24**, 703–714.
- Short, J., Turner, B., and Risbey, D.A. (2002). Control of feral cats for nature conservation. III. Trapping. *Wildlife Research* **29**, 475-487.
- Taraborelli, P.A., Dacar, M., and Gianonni, S.N. (2003). Effect of plant cover on seed removal by rodents in the Monte Desert (Mendoza, Argentina). *Austral Ecology* **28**, 651-657.
- Twyford, K.L., Humphrey, P.G., Nunn, R.P., and Willoughby, L. (2000). Eradication of feral cats (*Felis catus*) from Gabo Island, south-east Victoria. *Ecological Management and Restoration* **1**, 42–9.
- Van Polanen Petel, M. A., Marks, C.A., and Morgan, D.G. (2001). Bait palatability influences the caching behaviour of the red fox (*Vulpes vulpes*). *Wildlife Research* **28**, 395-401.



## 5. CHAPTER FIVE : INTERACTIONS BETWEEN A TOP ORDER PREDATOR AND EXOTIC MESOPREDATORS

Published as a research article in the International Journal of Ecology;

Moseby, K.E., Neilly, H., Read, J.L. and Crisp, H.A. (2012) Interactions between a top order predator and exotic mesopredators. International Journal of Ecology Article ID 250352, doi:10.1155/2012/250352.

Chapter 5- Moseby, X.E., Neilly, H., Read, J.R. and Crisp, H.A. (2012) Interactions between a top order predator and exotic mesopredators. International Journal of Ecology 2012 Article ID 250352, doi:10.1155/2012/250352.

**Author Contribution**

Conceived and designed the experiments- KEM, JLR

Performed the experiments including field work- KEM, HN, JLR, H-AC

Analysed the data- KEM

Contributed reagents/materials/analysis tools

Wrote the paper- KEM

**Author Declaration**

I approve the above author contribution statement and give permission for the paper/s co-authored by me to be included in the thesis by Katherine Moseby

---

Name: Heather Neilly

Signature:

Date:

Name: John L. Read

Signature:

Date:

Name: Helen A. Crisp

Signature:

Date:

## ABSTRACT

An increase in mesopredators caused by the removal of top order predators can have significant implications for threatened wildlife and their management. Recent evidence suggests that Australia's top order predator, the dingo, may suppress the introduced cat and red fox, although this relationship has not to date been trialled experimentally. Our study aimed to test this relationship by re-introducing 7 foxes and 6 feral cats into a 37 km<sup>2</sup> fenced paddock in arid South Australia inhabited by a male and female dingo. Radio-collars with GPS dataloggers recorded locations of all experimental animals every 2 hours. Interactions between species, mortality rates and post-mortems were used to determine the mechanisms of any direct or indirect suppression. Autopsy results, inspection of kill sites and GPS fix locations suggest that dingoes killed all 7 foxes within 17 days of fox introduction to the enclosure. No pre-death interactions were recorded between the dingoes and foxes. All 6 feral cats died between 20 and 103 days after release into the paddock. Autopsy results and GPS fix locations suggest that the dingoes were also implicated in the deaths of at least 3 of these feral cats. Dingoes typically stayed with fox and cat carcasses for several hours after death and/or returned to the carcass several times in ensuing days. There was no evidence of intraguild predation, interference competition was the dominant mechanism of suppression in our study. Results from this study support anecdotal evidence that dingoes may play a role in suppressing exotic medium-sized predators, particularly foxes, in arid environments. Whilst results suggest that dingoes could be used as a management tool in conservation programs, we outline further research programs required to determine if this suppression translates into a net benefit for different threatened prey species.

**KEY WORDS** Australia, competition, dingo, fox, cat

## 5.1 INTRODUCTION

Introduced feral cats (*Felis catus*) and red foxes (*Vulpes vulpes*) have been implicated in the historical extinction and decline of many Australian mammal species [1-4] as well as the failure of several recent attempts to reintroduce threatened species to the wild [5-9]. Effective control of the red fox and feral cat is a core objective of many Australian mammal and terrestrial bird recovery programs. Although the red fox has been successfully controlled in some areas of Australia using poison meat baits [11], the efficacy of long term baiting can attenuate due to high selection pressure for tolerance to 1080 [12] and bait shyness attributable to receiving a sub-lethal dose of poison. Control of foxes is also thought to lead to an increase in cat density [11,13,14] which could negate any positive biodiversity benefits. Poisoning feral cats is often ineffective owing to poor bait uptake [15-21] and a cost effective, large scale control mechanism for feral cats is currently not available [22].

Interspecific killing between carnivores is common [23], and recent studies have highlighted the possible role of top-order predators in controlling second-tier carnivores (mesopredators) [24,25,26]. The mesopredator release hypothesis predicts that reduced abundance of top-order predators results in increased abundance or activity of smaller subordinate predators [24]. This hypothesis has most support in North America, where studies have found that when coyote (*Canis latrans*) abundance declines, red fox numbers increase [24,27]. Removal

of the grey wolf, *Canis lupus*, has also been linked to an increase in coyote populations [28] and removal of coyotes has resulted in changes in bobcat (*Lynx rufus*) and gray fox (*Urocyon cinereoargenteus*) populations [29]. In Scandinavia, the pine marten (*Martes martes*) was found to increase after a decline in red fox populations [31].

Where predation efficiency or prey specificity of smaller predators is superior or different to that of the top order predator then changes in prey abundance can result [24]. Glen and Dickman [25] outlined complex interactions between carnivores in Australia and suggested that mesopredator release is an important mechanism shaping current prey populations in Australia. The post-European extinction of some Australian mammal species is thought to at least be partly attributable to mesopredator release through the removal or control of the dingo (*Canis lupus dingo*) [4]. Stable dingo populations are still found in many arid areas of Australia and may provide a net benefit to some threatened wildlife species through a decrease in predation rates by the red fox and/or feral cat [2,31-34]. Smith and Quin [2] found lower rates of conilurine rodent extinction in areas where dingoes were abundant, and Johnson *et al.* [4] has suggested that mammal extinctions and decline are less severe in areas where dingoes are still present. Letnic *et al.* [35,36] also favour the mesopredator release hypothesis as well as the trophic cascade theory, which suggests that top predators such as dingoes have either positive or negative effects on lower trophic levels and may indirectly enhance plant biomass [37]. The removal of dingoes may thus allow herbivores [38] and smaller introduced predators to increase, depleting plant biomass and increasing predation pressure.

Unfortunately little empirical data exist to support the perceived role of the dingo in suppressing fox and cat abundance at landscape scales [39] with evidence relying on correlations using historical or observational data [see 36,40]. However dingoes have been recorded occasionally killing or eating foxes [41] and cats [42,43] and remains of both have been recorded in dingo scats, although usually at a very low occurrence [41,44,45-47]. Dingoes are thought to exclude foxes from resource points such as carcasses during drought [46] and fox abundance has also been found to be higher in areas where dingoes are absent or controlled [32,36,45,48]. Dingoes could potentially suppress fox and cat populations through intraguild predation, interference and/or exploitative competition. Interference competition may include direct attack, exclusion from resource points, causing a change in habitat use or activity times or by increasing stress levels through frequent avoidance behaviour.

Dingoes are currently excluded or controlled over most of the Australian pastoral zone for the protection of commercial stock. Understanding any role that dingoes play in controlling introduced predators could assist in seeking a balance between the control of dingoes for pastoral production and the protection of dingoes for broader biodiversity benefits.

This study aimed to test the hypothesis that dingoes can suppress feral cats and foxes by examining their interactions within a landscape scale enclosure. A pair of dingoes was reintroduced to a 37 square km fenced paddock in northern South Australia. Feral cats and foxes were reintroduced 4 months later and all animals were monitored for up to 12 months using GPS datalogger collars. Interactions between species, mortality rates and post-mortems were used to determine if suppression was due to interference or exploitative competition and/or intraguild predation. Cats and foxes were also introduced to an adjacent unfenced control area where dingoes were removed. Indices of cat, fox and rabbit spoor were compared between the two areas. Two factors were critical to the study; firstly, that densities of dingoes, cats, foxes and prey species were typical of those found in the wider environment, and secondly that all study animals were local inhabitants and familiar with the habitats present in the study area.

## 5.2 STUDY AREA

A 37 km<sup>2</sup> ‘dingo paddock’ was fenced between July and November 2008 (30.27°S, 136.93°E) on Stuart Creek Pastoral Station. The paddock is situated approximately 35 km north of Roxby Downs in northern South Australia and is enclosed on three sides by a 1.6 m high netting fence (50mm holes) with a 50 cm floppy top curving inwards to keep dingoes, cats and foxes within the paddock but allowing cats and foxes to climb in. The netting fence was based on the Arid Recovery fence design [49] but was built from 50 mm netting to allow small rabbits to pass through the fence. The southern boundary of the paddock is shared with the Arid Recovery Reserve’s Red Lake enclosure and is a 1.15 m high netting fence made from 30 mm netting with a floppy top overhang facing the dingo paddock. This study was conducted between December 2008 and December 2009 and formed part of a larger predator behaviour study which began in January 2008.

The southern section of the dingo paddock comprised a clay interdunal swale more than 2 km wide and vegetated with chenopod shrubs, bladder saltbush (*Atriplex vesicaria*), Oodnadatta saltbush (*A. omissa*) and low bluebush (*Maireana astrotricha*). Longitudinal orange sand dunes supporting sandhill wattle (*Acacia ligulata*) and sticky hopbush (*Dodonaea viscosa*) shrublands were present in the northern sections, separated by 100 to 400 m wide swales. Other habitats include mulga (*Acacia aneura*) sandplains, patches of dune canegrass (*Zygochloa paradoxa*), and a breakaway range comprising silcrete capped hills with colourful eroding shale slopes in the western section of the paddock. Three ephemeral creeklines dissected the paddock from south to north and were characterised by denser vegetation cover and shallow sandy beds usually 1-2 m in width. Creeks flowed after rain into a near-permanent dam, a bulldozed depression in the soil located in the northern section of the paddock. The dam contained water throughout the study and water was also present at three minor pipeline leaks along the southern boundary.

We chose an unfenced control area south of the dingo fence, a man-made wire netting fence erected to exclude dingoes from southern sheep grazing areas. The control area was on adjoining Mulgaria Pastoral Station and situated 5 km east of the dingo paddock, a distance considered sufficient to ensure independence but close enough to contain similar habitat types and reflect similar climatic events. Habitats within the control area were similar to the dingo paddock with a large clay swale, an area of closely spaced sand dunes, a pastoral dam, and an area of breakaways. The dam within the control area was stocked with domestic cattle (*Bos taurus*).

The Roxby Down’s climate is arid, failing to reach its long term average rainfall of 166 mm in 60 % of years [50]. Rainfall is aseasonal and with equal likelihood of rain during any month. Productivity within arid zone ecosystems is driven by unpredictable rainfall events and only 100 mm of rainfall was recorded in both 2008 and 2009, leading to prolonged dry conditions. A significant rainfall event occurred just prior to the study in November 2008 (Fig. 1), which filled the dam within the paddock and led to a flush of grass growth. However, conditions then remained relatively dry until the end of the study period.

## 5.3 METHODS

### 5.3.1 DINGOES

In December 2008, a male and female dingo were captured from Stuart Creek Pastoral Station and released into the paddock. The wild dingoes were captured using soft catch Jake™ foot hold traps set around a cattle carcass located approximately 50 km north of the dingo paddock. Traps were fitted with springs to reduce injury and were checked in the late evening and again at dawn. No teeth damage was recorded after capture. We lightly anaesthetised the captured adult dingoes using a mixture of 1 ml of Metetomidine Hydrochloride and 0.5 ml of Ketamine, administered intramuscularly.

The anaesthetic was reversed using 0.5 ml of Atipamezole Hydrochloride. Anaesthetic and reversal doses for all animals were prepared in advance by a qualified veterinarian who also trained all animal handlers in correct administration of the pre-prepared doses. An anaesthesia procedure was developed, and approved by the Wildlife Ethics Committee, including monitoring of rectal temperature during anaesthesia. Dingoes were weighed, checked for reproductive condition and fitted with Global Positioning System (GPS) datalogger ARGOS satellite collars with VHF (SIRTRACK, Havelock, New Zealand) that nominally recorded fixes every 2 hours. Collars weighed 640 g and were no more than 4 % of dingo body weight, less than the manufacturer's and South Australian Wildlife Ethics Committee's maximum approved proportional collar weight of 5%. Dingoes were transported in an air-conditioned car and released at the dam within the dingo paddock on the same morning as capture. Dingoes were checked after two hours and were then radiotracked daily for the first week. Radiotracking fixes indicated that both dingoes began moving throughout the paddock within a few hours of release. Although the number of dingoes placed in the paddock mirrored regional density, we provided a food subsidy to determine whether the availability of prey was limited in the paddock and could have influenced study outcomes. Between December 2008 and October 2009 kangaroo or rabbit carcasses and occasionally meat offcuts were placed at least fortnightly at a carcass dump established near the dam within the dingo paddock. Two remote motion sensor cameras (DVR Eye, Pix Controller, PA, USA) were placed at the carcass dump to record activity.

Weekly ARGOS satellite downloads were used to check whether the dingoes were in the paddock and we conducted daily fence checks during the first month to repair any attempts to dig out under the fence. We recaptured the male and female dingoes in January and March 2010, respectively, to replace the GPS collars before the VHF batteries expired. No collar-related injuries such as rubbing or ulcerations were recorded. The male was captured using a single soft catch Jake™ trap set under an *Acacia ligulata* bush using a cat's head as bait, and the female was captured along the fenceline using a single Victor Soft-Catch™ (No. 1.5) trap. During the study, the pair of dingoes successfully raised a single male dingo pup born in June 2009. The female started using the breeding den in the northern sand dunes of the paddock on June 1 and continued to use it until July 16. After this time the female and pup moved around the paddock and frequently changed shelter sites.

### 5.3.2 CATS AND FOXES

Feral cats that remained in the paddock after construction were trapped in August 2008 and fitted with GPS data logger radiocollars with VHF (SIRTRACK, Havelock North, New Zealand) for a separate study comparing cat behaviour before and after dingo reintroduction. Cats were fitted with a small hind foot ring made from a cable tie with a 10cm length of light chain attached. The chain dragged behind the cat when it moved and left a small indentation in soft substrate where tracks could be detected indicating that the cat had been fitted with a radio-collar. All cat spoor recorded during quad bike traverses of the paddock immediately after trapping were from collared cats suggesting that most, if not all, cats within the paddock had been captured and radio-collared. Only two of these cats still remained alive when additional cats and foxes were placed in the paddock in April 2009 and by then foot rings had been removed.

Between April and October 2009, 4 to 10 months after the dingoes were released into the paddock, we captured six feral cats and seven foxes, fitted them with radiocollars and released them inside the dingo paddock. The majority of these cats and foxes were captured outside the paddock within 10 km of the dingo paddock in similar habitat (Table 1). However, one feral cat and one fox were captured inside the paddock, after the remote cameras detected that new animals had breached the fence and were visiting the carcass dump. The two radiocollared cats from a previous experiment that were resident in the paddock when the study began in April 2009 were also monitored during the study. Four of the seven foxes and all cats were captured in areas where dingo tracks are regularly observed suggesting they were not naive to dingo presence.

Animals were captured using Victor Soft-Catch™ (No. 1.5) rubber jawed leg-hold traps fitted with springs to prevent injury (Coast to Coast Vermin Traps). Two lures were used in association with the traps; 'pongo' (cat urine) and occasionally a Felid Attracting Phonic, 'FAP' (Westcare Electronics). Traps were checked in the evening or early each morning, and captured feral cats and foxes were restrained using gloves and towels, and anaesthetised with a mixture of Metetomidine Hydrochloride and Ketamine administered intramuscularly. It was not possible to weigh animals before sedation so doses were pre-prepared for small (less than 5 kg- 0.32 ml of Metetomidine and 0.2 ml of Ketamine) and large (more than 5 kg- 0.4 ml Metetomidine and 0.25 ml of Ketamine) cats and adult foxes (1 ml of Metetomidine and 0.5 ml of Ketamine).

We weighed and sexed the cats and foxes and noted the condition of their teeth, body and reproductive organs. Only animals weighing at least 2.7 kg were used in the study to ensure radiocollars remained less than 5 % of body weight (Table 1). The 135 g GPS data logger collars with VHF transmitter (SIRTRACK, Havelock North, New Zealand) were constructed from synthetic belting and recorded GPS fixes every 2 hours. The units were housed in epoxy resin and contained 2 antennas, micromouse GPS and 220 mm, 2NC gauge whip antenna. The VHF transmitter (40/80 ppm) was equipped with a mortality sensor, triggered after more than 24 hours without movement. The rectal temperature was taken every 3 minutes whilst under anaesthetic and cold packs were placed between the hind legs if the body temperature rose above 39 degrees Celsius. Animals were then given the reversal drug Atipamezole Hydrochloride, placed in a cage trap covered by a towel in a vehicle and only released when they had fully recovered. If the ambient temperature was over 30 degrees Celsius the towel was moistened and the vehicle air conditioned. We released animals at the dam or within sand dunes in the dingo paddock and watched to ensure they ran off after the cage trap was

opened. They were radiotracked either later that day or early the following day to ensure they had moved.

Between September and December 2009, we captured and radiocollared an additional three foxes and three feral cats and released them into the unfenced Mulgaria control area to act as controls. One control cat was trapped in the Mulgaria control area and the other two control cats were captured within 15 km of the control area. All control foxes were captured on Roxby Downs Station, 50 km south of the dingo paddock, two in October 2009 and one in December 2009.

Between April and December 2009, we radiotracked all collared animals within the dingo paddock and control area weekly or fortnightly on foot, quadbike or from a Cessna 172 aeroplane with a wing-mounted aerial. If an animal was found dead, its location was recorded and a thorough search of the death location ensued. Habitat, tracks, scats, bones, fur, warrens or any other signs of interest were recorded. Any fresh carcasses were sent to Zoos South Australia where post mortems were performed by qualified veterinarians.

### *5.3.3 DATA ANALYSIS*

We converted collar downloads from Greenwich Mean Time to Australian Central Standard Time (non daylight saving) and plotted them using Arc GIS software. Collar accuracy varied according to the number of satellites available at the time of the GPS fix but precision was usually less than 10 m. For deceased animals, GPS fix locations were used to confirm the point of death by identifying clusters of points in the same location indicating no movement for an extended period. The time of death was estimated as the time interval between the first GPS fix at the death location and the time of the last GPS fix recorded in an area prior to the death location, which typically permitted time of death to be estimated to within 2 hrs. In cases where multiple clusters of fixes were evident at a number of localities within a 1.5 km radius, ground searches revealed that carcasses had been dragged after death and the first cluster was identified as the kill site. Time and location of death of all cats and foxes within the paddock were compared to dingo GPS fix locations for the same period to determine whether the dingoes were present at the death location within the correct time interval. Other factors were also considered when determining the cause of death, including the results of any autopsies and presence of dingo tracks.

The distances between all fox and dingo GPS fix locations at each 2 hr interval were used to determine if any possible encounters had occurred between the two species prior to death. Given that the approximate dimensions of the paddock were 7 km by 5 km, distances of less than 500 m between animals within a 4 hr time interval were conservatively considered possible encounters. Additionally, all GPS fix locations within 24 hrs of death were closely compared to dingo locations to determine if the dingoes had followed the foxes prior to death. GPS fix locations of cats and dingoes were also compared but only for the 24 hr period prior to death as cats remained alive longer than foxes and produced significantly more GPS fix locations for analysis.

To investigate the influence of fox presence on dingo activity, each dingo's minimum daily distance moved was compared on days when foxes were present and absent in the paddock. Minimum daily distance was calculated as the total distance between successive GPS fix locations over a 24 hr period. At least one fox was in the paddock over three different periods between June and October for 34, 6 and 13 consecutive days respectively. Minimum daily



distances during these times were compared with the remaining 161 days when foxes were absent during the study period. Male and female dingoes were analysed separately using one way ANOVAs.

#### 5.3.4 PREY ABUNDANCE

Red kangaroos (*Macropus rufus*) remained present in both the dingo paddock and control area throughout the study. European rabbit (*Oryctolagus cuniculus*) warrens were common throughout the sandy dunes and sandplains and clusters of larger, more permanent rabbit warrens were located in calcrete outcrops throughout the clay swales. Warren systems of the spinifex hopping mouse (*Notomys alexis*) were present throughout the sand dunes, and, along with the Bolam's mouse (*Pseudomys bolami*) have consistently been the most common small mammal present on regional sand dunes over the preceding decade [51]. Other small mammals including the introduced house mouse (*Mus musculus*) and dunnarts, *Sminthopsis* spp., also occur in the region but at low densities, and are usually restricted to the clay interdunal swales [51].

Indices of dingo, fox, cat and rabbit activity were derived from the presence of spoor along 200 m track transects established in both the control and dingo paddock in the three main habitat types; sanddune, swale and creekline. Swale transects were all placed on roads where suitable substrate for tracking existed. Transects were swept clean using a metal bar dragged behind a quadbike the night before the first of two consecutive mornings of track counts. Data from the two mornings were combined to give a presence/absence score for each transect for each monitoring period. A total of 39 transects (20 sand dune, 10 creeklines and 9 swale) were established in the dingo paddock and 38 (20 dune, 8 creekline and 10 swale) in the control area. All transects were sampled every 4 months from February 2008 until February 2010. Sampling began 11 months prior to dingo reintroduction and continued for 3 months after the completion of the experiment.

## 5.4 RESULTS

Rainfall during the study period was erratic with the most significant rainfall event recorded in November 2008 (Fig. 1).

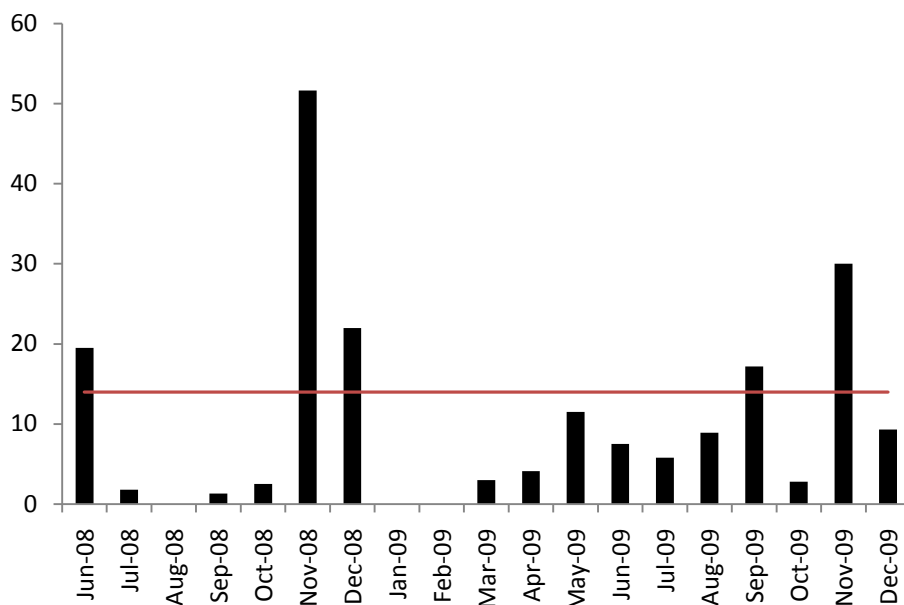


Figure 1: Rainfall recorded 6 months prior and during the study period. The line indicates the monthly average.

#### 5.4.1 FOXES

All seven foxes released into the dingo paddock died within 17 days of release (Table 1). GPS fix locations, kill site inspections and autopsies suggested that all seven animals were killed by dingoes. One fox appeared to have been killed by the female on her own when the male was at the den site. All other deaths occurred when the dingoes were travelling together. Where the time of death was known, foxes died between 10.30 pm and 3 am (Table 1). Four of the animals died on sand dunes and three on swales (Fig. 2). None of the deaths occurred in areas of dense vegetation. Deaths were recorded at various locations around the paddock with no apparent association with the breeding den or resource points (Fig. 2). Additionally, the deaths were recorded both during and after the female whelped. There was no indication that any of the foxes had been eaten and most exhibited little external sign of injury. Some carcasses were mauled and parts dragged up to 1500 m after death. In the four cases where the fox carcass was not retrieved for more than 12 hrs after death, the dingoes either remained with or returned to the carcass for up to 6 days after death (Table 2).

Three of the seven foxes (Fox 32, 36 and 37) were found within a few hours of death and could be necropsied (Table 2). Injuries sustained included ruptured leg muscles and/or trauma to the lumbar region and ribs with herniation of the abdominal muscles resulting in extensive and terminal haemorrhaging. Veterinarians from Zoos South Australia indicated that the injuries were consistent with an attack by dingo or dingoes. In one instance the fox had been chased several times at high speed around a bush. In another, scrape marks and diggings suggested the fox had been flushed out of a warren on a sand dune.

Tracks and GPS fix locations from the dingoes and foxes suggested that they also killed three other foxes (Table 2), with one or both dingoes recorded less than 10 m from the death points during the time of death. The remaining fox, Fox 31, was within 110 m of the male dingo

when it died. After death, the fox and male dingo GPS fix locations were within 10 m of each other at two different cluster locations up to 1.5 km from the kill site suggesting the carcass was dragged after death. Unfortunately the collar failed to record most of the female dingo fixes taken during the 17 h death period but she was travelling with the male just prior to the death period.

There were no recorded interactions between the foxes and dingoes prior to fox deaths. The only instance when dingo and fox fixes were recorded within 500 m of each other within a 4 h time interval was at the time of fox deaths. Furthermore, outside this 4 hr window, more than 450 m and 12 h were recorded between any fox and dingo locations suggesting that the first physical encounter between dingo and fox was also the last. There was also no indication that dingoes were following foxes prior to death as both species were moving in different directions and the distance between fox and dingo GPS fix locations recorded just prior to death was between 1703 and 3000 m (Table 2). No fox deaths were recorded along fencelines or roads despite both foxes and dingoes regularly using these features during the study. One fox collar did not store any fixes during its time in the dingo paddock so pre-death interactions with the dingoes could not be determined.

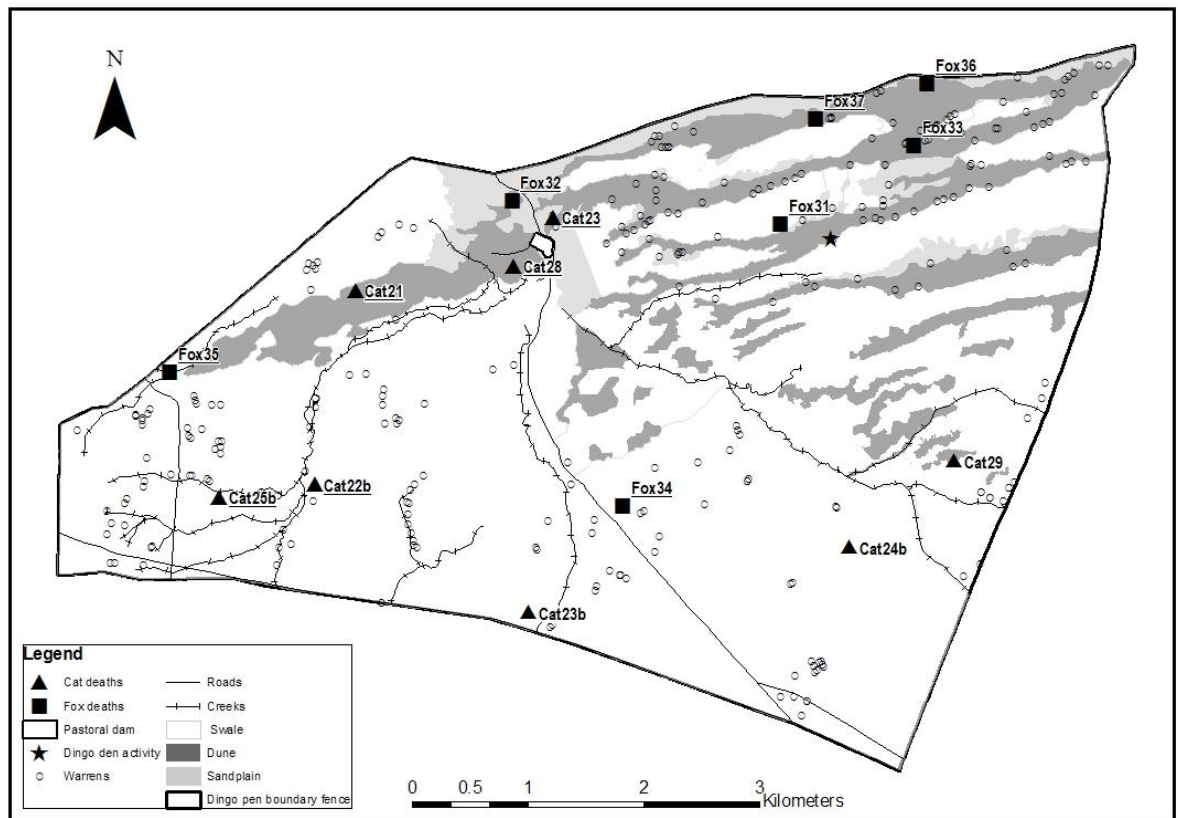


Figure 2: Location of animal deaths attributed to dingoes within the dingo pen. Habitat types, rabbit warrens and the dingo den site are also marked.

There was a strong trend towards longer daily movements in male and female dingoes when foxes were present in the paddock compared with when foxes were absent (female  $F=3.847$ ,  $df=1,213$   $P=0.051$ ; male  $F=3.434$ ,  $df=1,213$   $P=0.065$ ) but results were not significant. The average minimum daily distance moved by the female dingo increased from 2782 m to 3617 m when foxes were present in the paddock and the male average increased from 3375 m to 4267 m.

#### 5.4.2 CATS

All six feral cats released into the paddock died between 20 and 123 days after being translocated into the paddock and we recorded evidence that at least three cats were killed by dingoes. An additional two cats already present and radiocollared in the paddock when the experiment began also appeared to have been killed by dingoes. Where dingoes were implicated in deaths, three occurred in the early evening and one in the mid morning. When the female dingo killed two cats on her own, the male dingo was at the den site, more than 1 km from the death points. Deaths occurred before, during and after denning and were in different habitat types and locations around the paddock (Figure 2). Dingoes displayed similar post-death behaviour to that shown with killed foxes, staying with and/or returning to carcasses after death.

A post mortem confirmed death by dingo attack in one cat (cat 28, Table 2) but the 4 remaining cats were too decomposed for autopsy, so tracks, dingo behaviour and GPS fix locations were used to determine if the dingoes may have been involved in the cat deaths. Although in two instances (cat 25b and cat 23) the dingo fixes were several hundred metres from the cats during the death period, other factors such as direction of pre-death movement, post-death dingo behaviour and tracks and saliva marks suggested that the cause of death was dingo attack.

The cause of death could not be determined for three of the cats (Table 1). Cat 23b was several kilometres from the dingoes when it died out on a swale. Its remains were found under a wedge-tailed eagle (*Aquila audax*) nest suggesting it may have been killed or scavenged by an eagle. The other two cats were within 350 m and 400 m of the dingoes during the death period and it is possible that the dingoes were involved in these deaths. Of the five cats that remained in the paddock long enough to be recaptured and recollared during the study, two had lost weight, one had maintained weight and one had gained weight.

Collars were removed from dead cats and foxes and no rubbing or collar-induced injuries were detected. The dingoes were recaptured 12-18 months after initial capture and no collar injuries were detected.

Table 1: *Details, location and fate of cats and foxes captured and radiocollared during the experiment. Distance refers to how far away from the release area the animal was initially captured. Time of Death is the time interval between the first GPS fix recorded at the death location and the last fix recorded at a different location prior to death.*

<i>Animal</i>	<i>Distance from Paddock</i>	<i>Release Location</i>	<i>Sex</i>	<i>Release Wt</i>	<i>Release Date</i>	<i>Death Date</i>	<i>Days Alive</i>	<i>Time of Death</i>	<i>Habitat of death location</i>	<i>Cause of death</i>
Fox 31	5km	Paddock	M	3500	1/6/09	4/6/09	3	04:56-20:56	Swale	dingo
Fox 32	5km	Paddock	M	4500	9/6/09	13/6/09-14/6/09	5	08:30-08:30	Dune	dingo
Fox 33	In paddock	Paddock	F	3500	24/6/09	24/6/09-25/6/09	1	23.03-01.02	Dune	dingo
Fox 34	5km	Paddock	M	3500	27/6/09	12/7/09-13/7/09	17	22:31-00:31	Swale	dingo
Fox 35	50km	Paddock	F	5005	15/8/09	20/8/09	6	23:04-03:05	Swale	dingo
Fox 36	50km	Paddock	F	4400	18/10/09	28/10/09	10	22:54-02:54	Dune	dingo
Fox 37	50km	Paddock	F	5200	16/10/09	18/10/09-9/10/09	3	08:30-08:30	Dune	dingo
Cat 22b	In paddock	Paddock	M	4050	26/4/09	14/6/09-13/7/09	48-78	unknown	Swale	dingo
Cat 23b	7km	Paddock	F	2950	22/7/09	2/11/09	103	20:54-22:54	Swale	unknown
Cat 24b	10km	Paddock	F	3750	28/8/09	30/11/09-7/12/09	94-112	unknown	Swale	unknown
Cat 25b	10km	Paddock	F	4050	16/9/09	6/10/09	20	8.59-10.58	Creepline	dingo
Cat 28	10km	Paddock	F	2950	3/4/09	30/4/09	27	16:56-18:56	Dune	dingo
Cat 29	10km	Paddock	F	2750	26/4/09	30/5/09	34	19:19-07:18	Dune	unknown
Cat 21	In paddock	Paddock	F	2700	20/8/08	21/6/2009	300	17:04-19:04	Dune	dingo
Cat 23	In paddock	Paddock	M	4200	28/8/08	8/4/09	210	17:00-19:00	Dune	dingo
Fox 38	50km	Control	F	4400	18/10/09					fate unknown
Fox 39	50km	Control	M	4800	18/10/09					fate unknown
Fox 30	10km	Control	M	4000	14/12/09					fate unknown
Cat 27b	15km	Control	M	4650	3/10/09					fate unknown
Cat 28b	15km	Control	F	3950	31/10/09					fate unknown
Cat 26	In control	Control	M	3600	29/9/08	2/7/09	276			euthanased

Table 2: Details of animal deaths attributed to dingoes during the experiment. Evidence for dingo attack in bold. M=Male, F=Female

Animal	Dingo distance (m) from fox/cat during death period		At carcass hours after death	Distance from closest dingo at fix preceding kill	Both dingoes together prior to kill?	Dingo tracks at death site	Carcass dragged	Saliva on carcass	Autopsy confirmed dingo attack	Fox/cat movement in 2 hours prior to death (m)
	Male	Female								
Fox 31	<b>114</b>	1200 <sup>2</sup>	2-14,22-24,64,96-98,122-130 (M)	2014	Yes		<b>Yes</b>		n/a	100
Fox 32	<b>&lt;10</b>	470,680	48 (M)		Yes	<b>Yes</b>		<b>Yes</b>	<b>Yes</b>	
Fox 33	900 den	<b>&lt;10</b>	4.5-6.5 (F)	3000	No		<b>Yes</b>		n/a	218,230
Fox 34	<b>&lt;10</b>	<b>&lt;10</b>	2-16 <sup>1</sup> (MF)	2124	Yes			<b>Yes</b>	n/a	843
Fox 35	<b>&lt;10</b>	<b>&lt;10</b>	20-22 (MF)	1703	Yes	<b>Yes</b>			n/a	2833
Fox 36	<b>&lt;10</b>	<b>&lt;10</b>	2 <sup>1</sup> (F)	1668	Yes	<b>Yes</b>		<b>Yes</b>	<b>Yes</b>	1662
Fox 37	<b>110</b>	<b>110</b>	<sup>1</sup>		Yes	<b>Yes</b>		<b>Yes</b>	<b>Yes</b>	
Cat 21	3446 den	<b>&lt;10</b>	2,12,14,16 (F)	1885	No		<b>Yes</b>		n/a	574
Cat 22b	4878 den	<b>&lt;10</b>	2-6 (F)	2300	No				n/a	
Cat 23	<b>650</b>	<b>700</b>	26 (M)	800	Yes				n/a	270,670
Cat 25b	200	130	3-5 (M) 14-18 (F)	1726	Yes	<b>Yes</b>		<b>Yes</b>	n/a	1609
Cat 28	300	160	2-4,12,24 <sup>1</sup> (M) 2-4 (F)	1886	Yes	<b>Yes</b>	<b>Yes</b>		<b>Yes</b>	949

<sup>1</sup> body removed within 12-24 hours of death

<sup>2</sup> female collar failed to record 5 fixes during death period

### *5.4.3 CONTROL ANIMALS*

Only one cat and no foxes could be relocated after release into the control area. The cat that was captured within the Mulgaria control area remained in the control area for 276 days before it was recaptured and euthanased at the end of the experiment. This cat sheltered extensively in rabbit warrens on rocky swales and although usually staying within a 12 km linear area it was known to travel more than 35 km to the south and back again within a two week period. This cat was recaptured three times over the study and its weight remained between 3350 and 3600 g.

All other control animals were transferred to the control area from surrounding areas and despite more than five attempts to locate them using a light aircraft, they could not be found. Searches from the air included a 20 km radius around the control site, all of the original capture locations and 1km traverses across the control area. The fate of these animals remains unknown but it is likely that they moved away from the control area.

### *5.4.4 TRACK TRANSECTS*

Prior to and during fence construction, spoor, sightings and scats of wild dingoes, feral cats and foxes were all observed within the dingo paddock area. However, spoor counts and spotlighting transects indicated that there were no foxes or dingoes present in the paddock when the fence was completed. Subsequent spoor counts and remote cameras detected two uncollared foxes and two uncollared cats that had climbed into the paddock at different times during the experiment.

Both control and dingo paddock transects exhibited similar trends of cat activity during the initial stages of the project (Fig. 3). However, despite the presence of at least five cats in the paddock prior to dingoes being released in December 2008, as well as the addition of 4 cats in 2009 and another cat that was captured after climbing into the paddock, cat activity declined to zero by February 2010. All ten of these cats were radiocollared and all died during the experiment. Cat activity fluctuated in the control area but cats remained present throughout the experiment. Both areas experienced a decline in activity in 2009, possibly partly due to the dry conditions experienced during this time.

Fox spoor was recorded in the paddock when foxes were released in June but declined to zero by the end of the experiment. Fox activity in the control area was variable over the study period (Fig. 4). The presence of rabbit spoor on transects followed similar trends at both dingo and control sites and were recorded on 50 to 85% of track transects in the dingo and control areas during 2009 (Fig. 5). Inside the dingo paddock, dingo tracks were present on an average of 27% of track transects during the study period. The control area averaged dingo tracks on 4% of transects suggesting very low dingo activity.

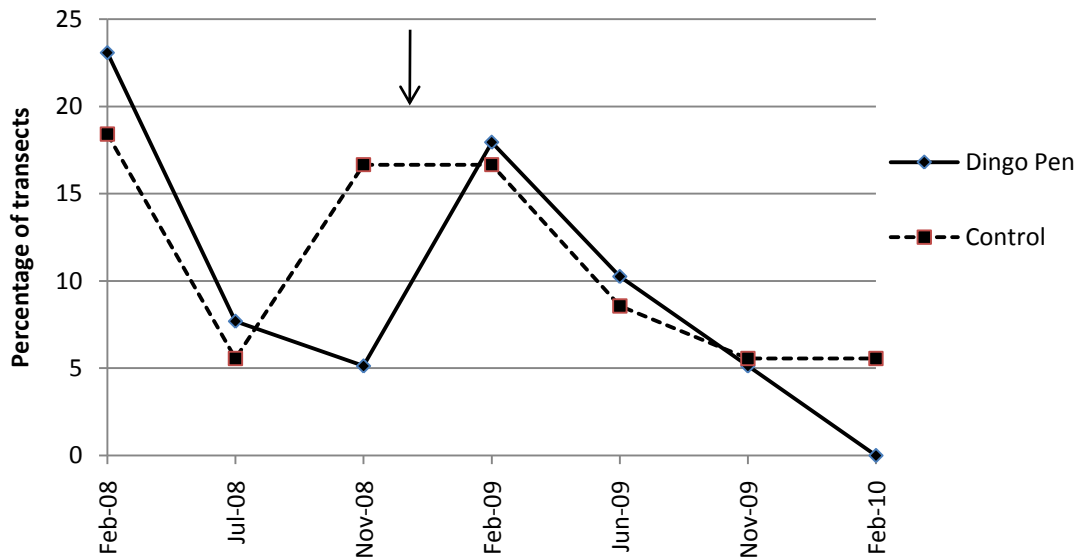


Figure 3: The percentage of transects (dingo pen n=39, control n=38) with cat tracks recorded at sites within the dingo pen and control area. Cats were added to the dingo pen between April and October 2009 and to the control area in October 2009. The pen was completed in November 2008 and the arrow indicates when dingoes were released.

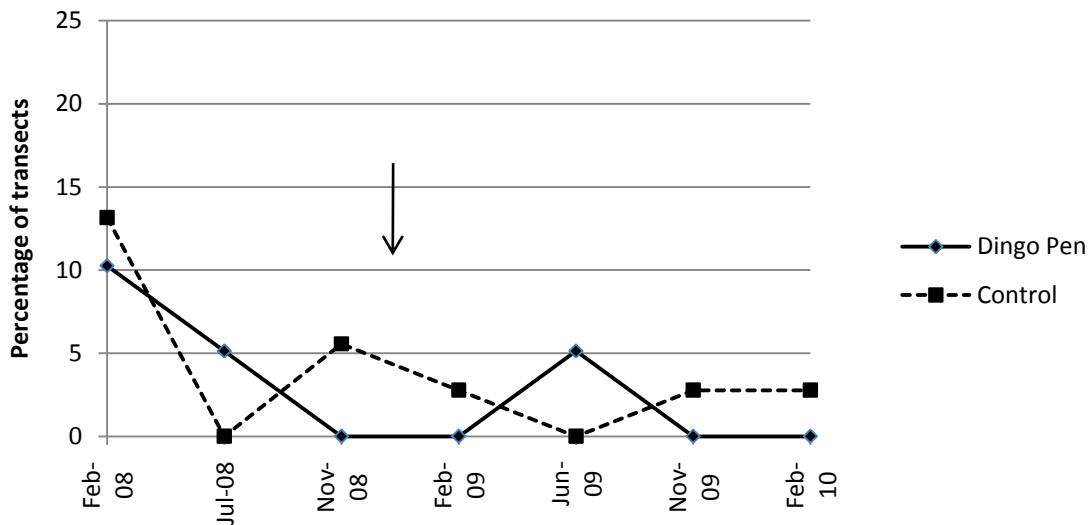


Figure 4: The percentage of transects (dingo pen n=29, control n=38) with fox tracks recorded at sites within the dingo pen and control area. Foxes were released into the dingo pen between June and October 2009 and into the control area between October and December 2009. The pen was completed in November 2008 and the arrow indicates when dingoes were released.



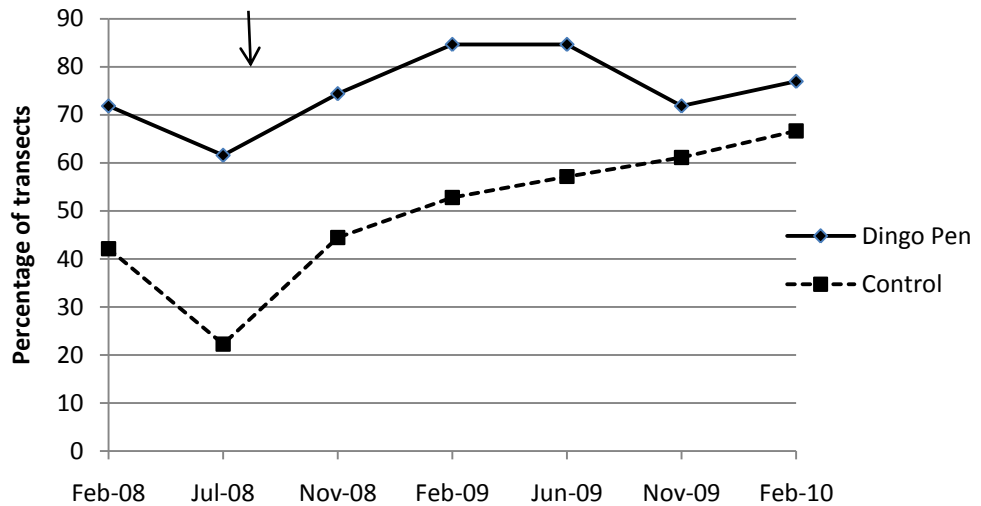


Figure 5. The percentage of transects (dingo pen n=39, control n=38) with rabbit tracks recorded within the dingo pen and control area. The arrow indicates when dingoes were released into the pen in December 2008.

## 5.5 DISCUSSION

Many previous studies have suggested that dingoes suppress fox abundance [2,4,35,36,37,38,48] but this is the first time that a direct negative interaction between dingoes and cats and foxes has been demonstrated. Small amounts of cat hair have been recorded in dingo scats [44] and some researchers have suggested that study cats were killed by dingoes [42]. However, other researchers have suggested that the presence of dingoes may assist cat survival by providing carrion [52]. Similar studies in North America have reported 25 % of radiocollared cats killed by coyotes [24]. Both male and female, and large and small, animals were killed by dingoes in our study suggesting that all foxes and cats may be susceptible to dingo attack.

The primary mechanism for suppression of cats and foxes by dingoes in this study appeared to be direct physical attack rather than suppression of breeding or exclusion from resource points as has been suggested elsewhere [46]. The dingoes did not eat any of the carcasses, despite staying with and/or returning to them for extended periods, which suggests they were killing due to interference competition rather than intraguild predation. Similar results were found by Molsher *et al.* [53] for red foxes and cats in Australia and Helldin *et al.* [54] for lynx (*Lynx lynx*) and red foxes in Sweden, radiocollared animals were killed but rarely eaten by the dominant predator. However, intraguild predation has been previously recorded in dingoes. Marsack and Campbell [41] observed dingoes eating foxes in arid Western Australia, and both fox and cat remains have been found in dingo scats and stomach contents [41,44-47]. Intraguild predation has also been recorded in the United States of America where cats are eaten by coyotes and can contribute up to 13.1% of coyote diet [55-57]. It is likely that mesopredator suppression mechanisms are influenced by resource availability, habitat type, breeding season and intraspecific behavioural differences. Interestingly, most animals killed by dingoes showed very little external sign of injury suggesting that many “unexplained”

deaths of radiocollared cats and foxes in other studies previously attributed to nutritional stress [e.g. 58] may have been the result of dingo attack.

Although cats were subjected to direct dingo attack, other forms of suppression may also have been occurring in the paddock. Burrows *et al.* [16] found higher breeding success of cats in the area from which foxes and dingoes had been controlled. Despite lactating cats being captured within the paddock in our study, and kitten spoor being found briefly at 1 cat den site, no successful cat recruitment was recorded. Juvenile cat spoor was not recorded on any track transects nor were any young uncollared cats photographed at the carcass dump. Other studies have suggested that dingoes may change cat's spatial behaviour, both Edwards *et al.* [58] and Palmer (pers. obs) found cats used wooded or mulga habitat more than open habitats when dingoes were present, possibly due to predation risk by dingoes. One cat in the paddock was found to frequently shelter in a wedge-tailed eagle nest in a mulga tree, one of the few trees in the paddock that was above 2 m in height. Cat deaths were recorded in all habitat types suggesting habitat may not have influenced predation risk in our study, but this result may not be consistent in wooded habitats.

Fox and cat deaths occurred at times when they were most active, foxes at night time and cats mainly at dusk. This is consistent with dingoes killing cats and foxes when they encounter them rather than digging them out of warrens or using olfactory cues to seek them out. Corroborating this assumption was the independent movement patterns of dingoes, cats and foxes in the 24 hrs prior to death and deaths occurring when animals unknowingly moved into the path of the dingoes or vice versa. Therefore it is likely that dingoes killed cats and foxes on an opportunistic basis but they were probably aware of the foxes in the paddock and may have increased their daily movements to increase the chances of encountering them.

Containing all three animal species within a paddock, albeit a landscape scale one, may have influenced the outcomes of the study by restricting the movement of some cats and foxes and perhaps rendering them more likely to encounter or be cornered by a dingo. The home range of cats and foxes varies considerably in the arid zone with averages of between 20 and 30 km<sup>2</sup> commonly recorded [16,21,58]. The average cat and fox home range recorded in the study area during a previous study was 16-17 km<sup>2</sup> with a range of 0.5 to 123 km<sup>2</sup> [21]. However, several factors suggest that the paddock represented a realistic arid zone environment. Although cat home ranges can be large, they do overlap [51] and track counts in control and paddock areas were similar at the start of the study, suggesting that the density of cats during the experiment was similar to that naturally recorded outside the paddock. Five cats were resident in the paddock when it was fenced, a similar or higher density to that maintained during the experiment. Only one fox was present in the paddock at a time and the density of 0.027 foxes per km<sup>2</sup> is much lower than that recorded in other arid zone studies (0.46-0.52 [59], 0.6 [60]). Rabbit track counts suggested that food resources were similar in control and paddock areas and the presence of a carcass dump provided supplementary food if required. Additionally, all animals used in the experiment were captured inside the paddock or from within 50 km in similar habitat.

The dingoes also appeared to have behaved typically, breeding in April/May and whelping in June, as recorded elsewhere in arid Australia [46]. The dingoes were recorded howling and scent marking and stopped trying to escape from the paddock after one week, also an indication that they were behaving like a dingo pack and maintaining a territory. Although dingo home ranges in the arid zone have been reported to be up to 77 km<sup>2</sup> [46] and even as large as 272 km<sup>2</sup> [61], other studies have reported arid zone dingo density between 1 and 22 per 100 km<sup>2</sup> (usually 5) [62], similar density to that recorded in the paddock during our study.

Pack size of two dingoes is commonly recorded in the arid zone [62]. Dingoes are known to feed on rabbits, reptiles, kangaroos and carrion [43], all of which were present during the study. Thus, although the paddock may have influenced the results, the large size, availability of different habitat types, densities of predators and presence of suitable food and rabbit warrens for shelter should have minimised these influences.

Deaths were recorded at various times between April and October, before, during and after the denning period. It is not known if dingoes will kill cats and foxes over the summer months. Although resident cats in the paddock when the dingoes were introduced in December were not killed until April or June, the dingo pair may have been more likely to influence other predators or competitors once they had formed a pack and started defending resources [63]. The female was only recorded killing animals on her own during her 6 week denning period when the male was guarding the den site. Dingoes may consider foxes and cats a threat to their pups and increase their intolerance of them during the breeding season. In North America, coyotes were also found to kill domestic cats at any time of year but with higher kill rates during pup-rearing [64].

Several animals appeared to have been chased around bushes or over short distances prior to death. The dingoes were travelling together when nine of the 12 foxes and cats were killed. It is impossible to determine if both dingoes assisted in the kills but it appears likely, as dingoes regularly hunt and kill prey cooperatively [65]. Additionally, tracking observations of a fox killed by dingoes in a separate arid site in South Australia indicated that 2 dingoes chased and killed a fox, whose fresh carcass was located on a sand dune (J.L. Read pers obs). The female dingo killed a large 4 kg cat and there was no indication that any particular size or sex of cat or fox was less susceptible to dingo attack by lone or pack dingoes. Cooperative dingo packs will more effectively hunt large mammals such as macropods, buffalo, feral horses or cattle [45,65,66] but solitary dingoes can effectively hunt rabbits, small mammals and sheep to achieve their daily energy requirements [43,45,46]. Grubbs and Krausman [64] documented coyotes killing domestic cats in the United States of America and found that single coyotes were just as effective at killing domestic cats as coyotes hunting in groups. However, dingoes may be more efficient hunters of cats as coyotes only killed cats in just over 50% of interactions [64].

Of those cats killed by dingoes, the resident cats survived longer than the cats placed in the paddock, possibly suggesting that the resident cats were more familiar with shelter sites and able to avoid interactions with the dingoes for longer. However, three of the five cats placed in the paddock do not appear to have been killed by dingoes and their causes of death are unknown. Feral cats in the arid zone are thought to suffer from periods of nutritional stress leading to high natural mortality of more than 50% in less than 12 months [21,58]. It appears unlikely that most cats were significantly nutritionally stressed as rabbit activity did not fluctuate significantly during the study period and carcasses were regularly dumped at the carcass dump but rarely used by cats. Additionally, most recaptured cats either maintained or increased in weight. All of the cats and foxes placed in the paddock were adults and had been previously surviving unaided in the paddock or surrounding similar habitat. The death of two of these resident cats also suggests that it is unlikely that the translocation itself was responsible for other cat deaths. Two of these cats may have been preyed on by wedge-tailed eagles but the third cat found down a rabbit warren may have died from natural causes.

Results from this study need to be extrapolated cautiously. Our experiment is a single replicate. Due to logistical constraints, we could only trial one pair of dingoes in a single paddock. Ideally, the experiment should be repeated using another dingo pair, and foxes and

cats added in different seasons. It is likely that interactions between cats, foxes and dingoes will vary depending on habitat types, breeding seasons and food availability. The relatively open habitat in the paddock, despite numerous rabbit warrens for shelter, may have made it easier for dingoes to locate and catch cats and foxes. More wooded environments or areas with denser understorey may enable cats, foxes and dingoes to co-exist more readily. Despite similar habitat types in capture and release locations, for some animals the paddock was an unfamiliar environment and may have influenced their susceptibility to dingo attack. Track searches of the paddock in early June 2010, 6 months after the experiment finished, located very low abundance of fox and cat tracks suggesting that these species had reinvaded the paddock. It is not known if dingoes permanently suppress cats and foxes over long periods or are more tolerant of cats and foxes outside the breeding period. Finally, drought conditions may have influenced results and increased dingo attacks due to competition for food resources.

Several studies have identified a loss in species biodiversity when a keystone or “apex” mammalian predator is removed [24,29,67]. The release of competitive restraints previously imposed on mesopredators can lead to changes in prey species’ composition and diversity. Previous research has suggested that dingoes may suppress cat and fox abundance but our trial is the first time that this has been proven experimentally. We found interference competition via direct attack to be the key suppression mechanism. However, the important question for threatened species conservation is whether the positive role that dingoes appear to play in suppressing cats and foxes will counteract dingo predation on these same threatened species and equate to a net benefit for native wildlife. We believe there are several critical factors that will determine whether a native species may benefit from cat and fox suppression. Firstly, the size and behaviour of prey species may be important. Medium-sized native mammals that are preyed on by cats and foxes and dingoes may not benefit to the extent of smaller mammals, for which dingoes are less efficient predators. Although dingoes are known to prey on smaller mammals such as rodents [39,43,46] they are not preferred stable prey items and may only be targeted during natural irruptions when they are plentiful. Solitary, wide-ranging species such as the Greater Bilby (*Macrotis lagotis*) may benefit more than communal sedentary species such as the Burrowing Bettong (*Bettongia lesueur*). Sedentary, communal species are more conspicuous and easier to target by predators. Proposed continued monitoring of rabbit and small native rodent abundance inside and outside the Arid Recovery dingo paddock should elucidate the net ecological role of dingoes for these different-sized mammals. Furthermore, reintroducing threatened native mammal species with different social and movement systems into the dingo paddock will help determine whether positive suppression of cats and foxes outweighs any direct predation by dingoes.

Secondly, like other canids, foxes and dingoes both have a predisposition to kill several prey and consume only few or none of the total kill [68]. This behaviour, known as surplus killing, is why dingoes and foxes can pose a significant threat to native fauna and sheep populations; especially spatially restricted or threatened populations [69,70]. There is some evidence to suggest that surplus killing in the dingo is not as common or devastating to native wildlife as the introduced red fox [68] but this is yet to be proven experimentally.

Thirdly, the relationship between dingo density and the magnitude of cat and fox suppression will have a major influence on whether a net benefit to prey species is realised. If low dingo density, particularly in concert with established breeding territories [63], is sufficient to significantly suppress cat and fox abundance then the net predation impact is likely to be low, leading to a net benefit to some wildlife species. However, the abundance of dingoes has

increased significantly since European settlement due to the proliferation of stock watering points. If the density of dingoes required to adequately suppress cats and foxes for the protection of wildlife is significantly higher than pre-European densities then any benefit to wildlife may be offset by artificially high predation rates by dingoes.

Finally, unlike cats and foxes, dingoes are dependent upon water, at least during summer. Therefore, in desert areas dingo density and their predation and mesopredator suppression will be spatially and temporally patchy compared with cats and foxes. Many desert animals rely on restricted refugia areas for survival during drought [71] and unless these refugia areas coincide with areas of mesopredator suppression, long term benefits to wildlife may not occur.

## 5.6 MANAGEMENT IMPLICATIONS

Although the ecological role of the dingo requires further verification in other environments, our study supports a growing body of evidence that the dingo plays an important role in ecosystem function. Therefore, we recommend that functional dingo populations in rangeland areas are maintained at landscape scales and that dingo control for calf protection is restricted to targeted control during exceptional circumstances. Research should now focus on whether dingoes provide a net benefit to threatened wildlife species by investigating the influence of prey size and behavioural traits, surplus killing and dingo density. We predict that smaller, solitary and wide-ranging native species close to permanent watering points will benefit the most from mesopredator suppression. Finally, the red fox, feral cat and dingo all have catholic diets that can change rapidly depending on resource availability. Despite the dingo arriving in Australia several thousand years ago, all three species are relatively new arrivals in Australia. Researchers should consider that the mechanisms and benefits of mesopredator suppression in Australia may not mirror those recorded in North America and Europe where mesopredators are usually native and their diets more prey-specific.

## 5.7 ACKNOWLEDGEMENTS

This study was conducted by Arid Recovery, a joint conservation initiative between BHP Billiton, the S.A. Department for Environment and Natural Resources, The University of Adelaide and the local community. Funding was provided by the South Australian Arid Lands Natural Resource Management Board, Olympic Dam Expansion Project and Arid Recovery. Greg Kammermann and his assistants worked tirelessly on building the dingo paddock on time and within budget. Many other volunteers and staff have contributed to this work particularly A. Kilpatrick, A. Clarke, T. Moyle, D. Sandow, C. McGoldrick, G. Miller and B. Arnold. Thanks to M. Lloyd for providing carcasses for our carcass dump and assisting with fox captures. Rob Savage gave permission to use Mulgaria Station as a control site and Dr D. Paton provided useful comments on the manuscript. We are indebted to the veterinarians from Zoos South Australia, in particular I. Smith, W. Boardman and D. McLelland, who conducted the necropsies, provided veterinary advice and assisted with attempting to retrieve the dingo collars. Dr A. Melville-Smith from the Roxby Downs Veterinary Clinic assisted with anaesthesia dosages and training. This study was conducted

under ethics approval (permit no. 6/2007-M3) from the South Australian Wildlife Ethics Committee.

## 5.8 REFERENCES

1. **Seebeck, J.H.** 1979. Status of the barred bandicoot *Perameles gunnii*, in Victoria with a note on husbandry of a captive colony. *Australian Wildlife Research*, **6**, 255-64.
2. **Smith, A.P., & Quin, D.J.** 1996. Patterns and causes of extinction and decline in Australian conilurine rodents. *Biological Conservation*, **77**, 243-267.
3. **Short, J.** 1998. The extinction of rat-kangaroos (Marsupialia:Potoroidae) in New South Wales, Australia. *Biological Conservation*, **86**, 365–377.
4. **Johnson, C.N., Isaac, J.L. & Fisher, D.O.** 2007. Rarity of a top predator triggers continent-wide collapse of mammal prey: dingoes and marsupials in Australia. *Proceedings of the Royal Society Biological Sciences*, **274**, 341-346  
doi:10.1098/rspb.2006.3711
5. **Short, J., Bradshaw, S.D. Prince, R.I.T. & Wilson, G.R.** 1992. Reintroduction of macropods (Marsupialia: Macropodidae) in Australia - A review. *Biological Conservation*, **62**, 189-204.
6. **Christensen, P. E. S., & Burrows, N.D.** 1995. Project Desert Dreaming: the reintroduction of mammals to the Gibson Desert. In: *Reintroduction Biology of Australian and New Zealand Fauna* (Ed by M. Serena), pp 199-208. Surrey Beatty and Sons: Chipping Norton, New Zealand.
7. **Gibson, D. F., Lundie-Jenkins, G., Langford, D.G., Cole, J.R. , Clarke, D.E. & Johnson K.A.** 1994. Predation by feral cats, *Felis catus*, on the rufous hare-wallaby, *Lagorchestes hirsutus*, in the Tanami Desert. *Australian Mammalogy*, **17**, 103-7.
8. **Southgate, R., & Possingham, H.** 1995. Modelling the reintroduction of the greater bilby *Macrotis lagotis* using the metapopulation model analysis of the likelihood of extinction. *Biological Conservation*, **73**, 151-60.
9. **Priddel, D., & Wheeler, R.** 2004. An experimental translocation of brush-tailed bettongs (*Bettongia penicillata*) to western New South Wales. *Wildlife Research*, **31**, 421-432.
10. **Thomson, P. C., and Algar, D.** 2000. The uptake of dried meat baits by foxes and investigations of baiting rates in Western Australia. *Wildlife Research*, **27**, 451–456.  
doi: 10.1071/WR99034
11. **Algar, D., & Smith, R.** 1998. Approaching Eden. *Landscape*, **13**, 28-34.
12. **Twigg, L.E., Martin, G.R. & Low, T.J.** 2002. Evidence of pesticide resistance in medium-sized mammalian pests: a case study with 1080 poison and Australian rabbits. *Journal of Applied Ecology*, **39**, 549-560.
13. **Risbey, D.A., Calver, M.C. & Short, J.** 1999. The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. I. Exploring potential impact using diet analysis. *Wildlife Research*, **26**, 621-630.
14. **Catling, B., & Reid, A.M.** 2003. Predator and critical weight range species 5. Results of Spring 2002 and Autumn 2003 surveys. CSIRO Sustainable Ecosystems, Canberra, Australia.
15. **Risbey, D. A., Calver, M.C. & Short, J.** 1997. Control of feral cats for nature conservation. I. Field tests of four baiting methods. *Wildlife Research*, **24**, 319–326.  
doi: 10.1071/WR96051

16. **Burrows, N. D., Algar, D., Robinson, A.D., Sinagara, J., Ward, B. & Liddelow, G.** 2003. Controlling introduced predators in the Gibson Desert of Western Australia. *Journal of Arid Environments*, **55**, 691–713. doi: 10.1016/S0140-1963(02)00317-8
17. **Algar, D., & Burrows, N.D.** 2004. Feral cat control research: Western Shield review-February 2003. *Conservation Science Western Australia*, **5**, 131-163.
18. **Hegglin, D., Bontadina, F., Gloor, S., Swild, J.R., Muller, U., Breitenmoser, U. & Deplazes P.** 2004. Baiting red foxes in an urban area: a camera trap study. *Journal of Wildlife Management*, **68**, 1010–1017.
19. **Olsson, M., Wapstra, E. Swan, G. Snaith, E. Clarke, R. & Madsen, T.** 2005. Effects of long-term fox baiting on species composition and abundance in an Australian lizard community. *Austral Ecology*, **30**, 899–905. doi: 10.1111/j.1442-9993.2005.01534.x
20. **Algar, D., Angus, J.G. Williams, M.R. & Mellican, A.E.** 2007. Influence of bait type, weather and prey abundance on bait uptake by feral cats (*Felis catus*) on Peron Peninsula, Western Australia. *Conservation Science Western Australia*, **6**, 109-149.
21. **Moseby, K.E., Stott, J. & Crisp, H.** 2009. Improving the effectiveness of poison baiting for the feral cat and European fox in northern South Australia: The influence of movement, habitat use and activity. *Wildlife Research*, **36**, 1-14.
22. **Denny, E.A., & Dickman, C.R.** 2010. Review of Cat Ecology and Management Strategies in Australia, Invasive Animals Cooperative Research Centre: Canberra, Australia.
23. **Palomares, F. & Caro, T.M.** 1999. Interspecific killing among mammalian carnivores. *American Naturalist*, **153**, 492–508.
24. **Crooks, K.R., & Soulè, M.E.** 1999. Mesopredator release and avifaunal extinctions in a fragmented system. *Nature*, **400**, 563-566.
25. **Glen, A.S. and Dickman, C. R.** 2005. Complex interactions among mammalian carnivores in Australia, and their implications for wildlife management. *Biological Review*, **80**, 387-401
26. **Ritchie, E.G. & Johnson, C.N.** 2009. Predator interactions, mesopredator release and biodiversity conservation. *Ecology Letters*, **12**, 982-998.
27. **Goodrich, J.M., & Buskirk, S.W.** 1995. Control of abundant native vertebrates for conservation of endangered species. *Conservation Biology*, **9**, 1357-64.
28. **Peterson, R. O.** 1995. Wolves as interspecific competitors in canid ecology. In: *Ecology and conservation of wolves in a changing world*. (L.N. Carbyn, S.H. Fritts and D.R. Seip) pp 315-323. Canadian Circumpolar Institute: Alberta, Canada.
29. **Henke, S. E. & Bryant, F.C.** 1999. Effects of coyote removal on faunal community in western Texas. *Journal of Wildlife Management*, **63**, 1066–1081.
30. **Lindstrom, E. R., Brainerd, S.M. Helldin, J.O. & Overskaug, K.** 1995. Pine marten–red fox interactions: a case of intraguild predation? *Annales Zoologici Fennici*, **32**, 123–130.
31. **Hobbs, R.J.** 2001. Synergisms among Habitat Fragmentation, Livestock Grazing, and Biotic Invasions in Southwestern Australia. *Conservation Biology*, **15**, 1522-1528.
32. **Newsome, A. E.** 2001. The biology and ecology of the dingo. In: *A symposium on the dingo*. (Eds C.R. Dickman and D. Lunney) pp 20-23. Royal Zoological Society of New South Wales: Sydney, Australia.
33. **Newsome, A.E., Catling, P.C. Cooke, B.D. & Smyth, R.** 2001. Two ecological universes separated by the dingo fence in semi-arid Australia: interactions between landscapes, herbivory and carnivory, with and without dingoes. *Rangeland Journal*, **23**, 71–98.

34. Daniels, M.J. & Corbett, L. 2003. Redefining introgressed protected mammals; when is a wildcat a wild cat and a dingo a wild dog? *Wildlife Research*, **30**, 213-18.
35. Letnic, M., Koch, F. Gordon, C. Crowther, M.S. & Dickman, C.R. 2000. Keystone effects of an alien top-predator stem extinctions of native mammals. *Proceedings of the Royal Society-Biological Sciences*, **276**, 3249-3256.
36. Letnic, M., Crowther, M.S. & Koch, F. 2009. Does a top-order predator provide an endangered rodent with refuge from an invasive mesopredator? *Animal Conservation*, **2009**, 1-11.
37. Hairston, N. G., Smith, F.E. & Slobodkin, L.B. 1960. Community structure, population control, and competition. *American Naturalist*, **95**, 421-425.
38. Pople, A.R., Grigg, G.C. Cairns, S.C. Beard, L.A. & Alexander, P. 2000. Trends in the numbers of red kangaroos and emus on either side of the South Australian dingo fence: Evidence for predator regulation? *Wildlife Research*, **27**, 269-76.
39. Mitchell, B.D., & Banks P.B. 2005. Do wild dogs exclude foxes? Evidence for competition from dietary and spatial overlaps. *Austral Ecology*, **30**, 581-591.
40. Wallach, A., Johnson, C.N., Ritchie, E.G. & O'Neill, A.J. 2010. Predator control promotes invasive dominated ecological states. *Ecology Letters*, **13**, 1008-1012.
41. Marsack, P., & Campbell, G. 1990. Feeding behavior and diet of dingoes in the Nullabor region, Western Australia. *Wildlife Research*, **17**, 349-57.
42. Palmer, R. 1996. Feral Pest Program Project 12, Cat research and management in Queensland, year 4; and Project 43. Dispersal and spatial organisation of cats on the Diamantina Plains, year 3. Interim Final Report to the Feral Pests Program, Australian Nature Conservation Agency, Canberra, Australia.
43. Fleming, P., Corbett, L., Harden, R. & Thomson, P. 2001. Managing the impacts of dingoes and other wild dogs. Bureau of Rural Sciences: Canberra, Australia.
44. Lunney, D., Trigg, B. Eby, P. & Ashby, E. 1990. Analysis of scats of dogs, *Canis familiaris*, and foxes, *Vulpes vulpes* (Canidae Carnivora), in coastal Bega, New South Wales. *Australian Wildlife Research*, **17**, 61-68.
45. Thomson, P.C. 1992 The behavioural ecology of dingoes in north-western Australia. III. hunting and feeding behaviour and diet. *Wildlife Research*, **19**, 531-41.
46. Corbett, L. 1995. *The Dingo in Australia and Asia*. University of New South Wales Press: Sydney, Australia
47. Paltridge, R.M. 2002. The diet of cats, foxes and dingoes in relation to prey availability in the Tanami Desert, Northern Territory. *Wildlife Research*, **29**, 389-403.
48. Johnson, C. N., & Van Der Wal, J. 2009. Evidence that dingoes limit abundance of a mesopredator in eastern Australian forests. *Journal of Applied Ecology*, **46**, 641–646. (doi:10. 1111/j.1365-2664.2009.01650.x)
49. Moseby, K.E. & Read, J.L. 2006. Moseby, K.E. and Read, J.L. (2006). The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biological Conservation*, **127**, 429-437.
50. Read, J. 1995. Recruitment characteristics of the white cypress pine (*Callitris glaucophylla*) in arid South Australia. *The Rangeland Journal*, **17**, 228–240. doi: 10.1071/RJ9950228
51. Moseby, K.E., Hill, B.M. & Read, J.L. 2009. Arid Recovery- A comparison of reptile and small mammal populations inside and outside a large rabbit, cat and fox-proof enclosure in arid South Australia. *Austral Ecology*, **34**, 156-169.
52. Paltridge, R., Gibson, D. & Edwards, G. 1997. Diet of the feral cat (*Felis catus*) in central Australia. *Wildlife Research*, **24**, 67-76.



53. **Molsher, R., Newsome, A. and Dickman, C.** 1999. Feeding ecology and population dynamics of the feral cat (*Felis catus*) in relation to the availability of prey in central-eastern New South Wales. *Wildlife Research*, **26**, 593-607.
54. **Helldin, J.O., Liberg, O. & Gloersen, G.** 2006. Lynx (*Lynx lynx*) killing red foxes (*Vulpes vulpes*) in boreal Sweden—frequency and population effects. *Journal of Zoology*, **270**, 657-663.
55. **MacCracken, J.G.** 1982. Coyote food in a Southern California suburb. *Wildlife Society Bulletin*, **10**, 280–281.
56. **Shargo, E. S.** 1988. Home range, movement, and activity patterns of coyotes (*Canis latrans*) in Los Angeles suburbs. Dissertation, University of California, Los Angeles, USA.
57. **Quinn, T.** 1997. Coyote (*Canis latrans*) food habits in three urban habitat types of western Washington. *Northwest Science*, **71**, 1–5.
58. **Edwards, G. P., De Preu, N.D., Shakeshaft, B.J., Crealy, I.V. & Paltridge, R.M.** 2001. Home range and movements of male feral cats (*Felis catus*) in a semiarid woodland environment in central Australia. *Austral Ecology*, **26**, 93–101. doi: 10.1046/j.1442-9993.2001.01091.x
59. **Marlow, N.J., Thomson, P.C., Algar, D. Rose, K. Kok, N.E. & Sinagra, J.A.** 2000. Demographic characteristics and social organisation of a population of red foxes in a rangeland area in Western Australia. *Wildlife Research*, **27**, 457-64.
60. **Read, J. L., & Bowen, Z.** 2001. Population dynamics, diet and aspects of the biology of feral cats and foxes in arid South Australia. *Wildlife Research*, **28**, 195–203. doi: 10.1071/WR99065
61. **Eldridge, S.R., Shakeshaft, B.J. & Nano. T.J.** 2002. The impact of wild dog control on cattle, native and introduced herbivores and introduced predators in central Australia. Final Scientific Report to Bureau of Rural Sciences, Northern Territory, Australia.
62. **Thomson, P.C.** 1992. The behavioural ecology of dingoes in north-western Australia. IV. social and spatial organisation, and movements. *Wildlife Research*, **19**, 543-63.
63. **Wallach, A.D., Ritchie, E.G., Read, J & O'Neill, A.J.** 2009. More than mere numbers: the impact of lethal control on the social stability of a top-order predator. *PLoS ONE*, **4**, e6861. doi:10.1371/journal.pone.0006861
64. **Grubbs, S.E. & Krausman, P.R.** 2009. Observations of coyote-cat interactions. *Journal of Wildlife Management*, **73**, 683–685.
65. **Corbett, L. K., & Newsome, A.E.** 1987. The feeding ecology of the dingo. III. Dietary relationships with widely fluctuating prey populations in arid Australia: an hypothesis of alternation of predation *Oecologia*, **74**, 215–227
66. **Newsome, A. E., & Coman B.J.** 1989. Canids. In: *Fauna of Australia. Mammalia Vol. 1B* (Eds D.W. Walron and B.J. Richardson), pp 993-1005. Australian Government Publishing Service: Canberra, Australia.
67. **Soulé ME, Estes JA, Miller B, Honnold DL.** 2005. Strongly interacting species: Conservation policy, management, and ethics. *BioScience* **55**, 168–176.
68. **Short, J., Kinnear, J.E. and Robley, A.** 2002. Surplus killing by introduced predators in Australia—evidence for ineffective anti-predator adaptations in native prey species? *Biological Conservation*, **103**, 283-301.
69. **Kinnear, J.E., Onus, M.L. & Bromilow, R.N.** 1988. Fox control and rock-wallaby population dynamics. *Australian Wildlife Research*, **15**, 435-50.
70. **Kinnear, J.E., Onus, M.L. & Sumner N.R.** 1998. Fox control and rock-wallaby population dynamics - II. an update. *Wildlife Research*, **25**, 81-8.

- 71. Morton, S.R.** 1990. The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia*, **16**, 201-213.

## 6. CHAPTER SIX: CAN PREDATOR AVOIDANCE TRAINING IMPROVE REINTRODUCTION OUTCOMES FOR THE BILBY?

Published as a research article in the international journal *Animal Behaviour*;

Moseby, K.E., Cameron, A. and Crisp, H.A. (2012). Can predator avoidance training improve reintroduction outcomes for the Bilby (*Macrotis lagotis*) in arid Australia? *Animal Behaviour* 83(4) 1011-1021.

Chapter 6- Moseby, K.E., Cameron, A. and Crisp, H.A. (2012). Can predator avoidance training improve reintroduction outcomes for the Bilby (*Macrotis lagotis*) in arid Australia. *Animal Behaviour*. 10.1016/j.anbehav.2012.01.023

**Author Contribution**

Conceived and designed the experiments- KEM,HAC

Performed the experiments including field work-KEM, AC,HAC

Analysed the data-KEM

Contributed reagents/materials/analysis tools

Wrote the paper-KEM

**Author Declaration**

I approve the above author contribution statement and give permission for the paper/s co-authored by me to be included in the thesis by Katherine Moseby

---

Name: Amber Cameron

Signature:

Date: \_\_\_\_\_

Name: Helen A. Crisp

Signature:

Date: \_\_\_\_\_

## ABSTRACT

Many threatened species reintroductions in Australia fail due to predation by introduced cats and foxes. We attempted to improve reintroduction outcomes by training greater bilbies to recognise cats as predators. The movement and behaviour of trained and control bilbies was compared in both a predator-free environment and in an area where cats and foxes were present. Trained bilbies within the predator-free environment moved significantly further, used more burrows with more entrances and changed burrows more frequently than untrained control animals. Trained bilbies also moved burrows when presented with olfactory predator stimuli whilst control bilbies did not. However, when bilbies were reintroduced to an area where predators were present, there was no difference in survival, movement or burrow use between trained and control bilbies. Both groups exhibited high survival rates in the first six months after release. In the presence of predators, both trained and control bilbies appeared to be more predator-aware possibly due to control bilbies learning from trained conspecifics or co-habiting rabbits or through inadvertent contamination of control animals during training. Results suggest that bilbies can be taught to recognise exotic predators but this may not necessarily translate into improved reintroduction success in the wild. Further investigation into cultural and filial transfer of anti-predator behaviour is recommended.

## 6.1 INTRODUCTION

Predation by the introduced feral cat (*Felis catus*) and red fox (*Vulpes vulpes*) has been responsible for the failure of many mammal reintroduction programs in Australia (Fischer and Lindenmayer 2000; Short 2009; Moseby *et al.* 2011). The reason for high predation rates by introduced predators on native prey can often be attributed to prey naivety, where a species does not express an effective anti-predator behaviour (Cox and Lima 2006). Banks and Dickman (2007) suggest there are three levels of prey naivety to alien predators: level 1 is a failure of prey to recognise a species as a predator; level 2 is recognition of the predator but adoption of inappropriate antipredator behaviour; and level 3 is where prey recognise the predator, have an appropriate response but the predator exhibits superior hunting skills. Prey naivety can occur due to isolation of individuals from predators during their lifetime (ontogenetic naivety) or, as is most likely in Australia (Griffin *et al.* 2000), through an absence of co-evolution (evolutionary naivety). To date, strategies to address prey naivety in Australia have largely concentrated on removing or controlling the predator itself rather than improving predator avoidance strategies in prey.

Lack of predator recognition (level 1 naivety) is thought to be the most damaging form of naivety (Cox and Lima 2006) and has been demonstrated in some Australian studies where prey species failed to recognise the introduced red fox as a predator (Banks 1998, Blumstein *et al.* 2002). Some researchers have attempted to improve recognition by using predator avoidance training where predator cues are paired with an unpleasant experience. Empirical evidence suggests a range of taxa including fish (Magurran 1989), birds (McLean *et al.* 1999) and mammals (Mineka and Cooke 1988; Griffin *et al.* 2002; McLean *et al.* 2000) can be trained to improve their antipredator skills. Griffin *et al.* (2002) found that the tamar wallaby (*Macropus eugenii*) could be trained to respond to a model fox, and rufous hare-wallabies (*Lagorchestes hirsutus*) were conditioned to fear a model fox after its presence was paired with a loud noise and wallaby alarm calls or squirts from a water pistol (McLean *et al.* 1996). Captive raised siberian polecats (*Mustela eversmanni*) showed increased alert

behaviour after the presence of a model owl and badger were paired with an aversion event (animals shot with elastic bands), with individuals reacting fearfully after just one training event (Miller *et al.* 1990). Some attempt was made to expose burrowing bettongs to dogs to reduce their naivety at Heirisson Prong in the early years of the reintroduction but this was later abandoned when more effective predator control was implemented (Short and Turner 2000).

Level 1 (Lack of recognition) ontogenetic naivety is arguably the easiest to address using classical conditioning as the species is likely to have retained some innate predator avoidance behaviour (Griffin *et al.* 2000), and once trained to recognise a predator may be able to implement an appropriate response. Training animals with evolutionary naivety is much more difficult unless they have co-evolved with a similar predator to the one which the species is being trained to avoid (Griffin *et al.* 2000). Similarly, level 3 naivety, where the predator exhibits superior hunting skills, is unlikely to be improved using classical predator-avoidance training as the animal already recognises and responds appropriately to predators.

Many prey animals monitor predator odour to reduce their risk of predation (Kats and Dill 1998; Perot-Sinal *et al.* 1999) often avoiding areas where predator scent is located (Sullivan and Crump 1984). Researchers have used predator odour to mimic predation risk under laboratory conditions and found that rodents have responded to mustelid predator odours by reducing mobility, shifting the timing of activity, selection of different microhabitats and even breeding suppression (Ylönen *et al.* 2001). Research into the use of olfactory cues in predator avoidance training has shown that in most studies, predator-experienced species changed their behaviour but predator-naive species did not (Blumstein *et al.* 2002). Species with ontogenetic or evolutionary naivety may have to learn to recognise the olfactory cues of predators (Blumstein *et al.* 2002) and predator odour may be a useful tool in predator-avoidance training.

Although several studies have demonstrated that predator-avoidance training can lead to a change in behaviour of the prey animal, few studies have investigated its effectiveness in field trials (for two examples see Ellis *et al.* 1977; Miller *et al.* 1990). In order to be an effective reintroduction tool, the learned behaviour needs to be retained by released animals, ideally reinforced soon after release and transferred to offspring. We investigated predator avoidance training using greater bilbies (*Macrotis lagotis*) that had been previously reintroduced to the predator-free Arid Recovery Reserve in South Australia. A previous attempt to reintroduce this species outside the Reserve in 2004 failed due to predation by feral cats (Moseby *et al.* 2011). The aims of the study were three fold. First, to determine if predator-naive bilbies could be trained to respond to predator cues by associating them with an unpleasant experience. Second, to determine what types of behavioural responses were triggered by antipredator-training and third, whether these responses translated to effective predator avoidance strategies in the wild. Predator avoidance trials were conducted in two stages. Initially, greater bilbies were exposed to predator training and control treatments within the Arid Recovery Reserve where predators were absent. To quantify the effects of predator avoidance training, movement behaviour of untrained control bilbies was compared with trained animals. We hypothesised that trained animals would relocate from their burrows when predator scent was added but untrained animals would not. Furthermore, we expected that trained bilbies would exhibit increased vigilance compared with untrained bilbies, manifested through differences in movement patterns. The second stage of the study involved a release of trained and untrained Arid Recovery bilbies outside the Arid Recovery Reserve where cats and foxes were present in low densities. We compared survival and post-release behaviour of trained and untrained bilbies and hypothesised that trained bilbies would exhibit

higher survival rates than untrained bilbies. We used results from this study to suggest the type of predator naivety expressed by captive-bred bilbies and whether predator-training would improve reintroduction outcomes.

## 6.2 METHODS

### 6.2.1 STAGE ONE- PREDATOR FREE ENVIRONMENT

#### **6.2.1.1 Study Site**

Established in 1997, the Arid Recovery Reserve (30°29'S, 136°53'E) is a 123 km<sup>2</sup> fenced enclosure situated 20 km north of Roxby Downs in arid South Australia (Fig. 1). The Reserve is divided into paddocks and rabbits, cats and foxes have so far been removed from 60 square kilometres of the reserve (Fig.1). Populations of four locally extinct threatened species, the greater bilby, greater stick-nest rat (*Leporillus conditor*), burrowing bettong (*Bettongia lesueur*) and western barred bandicoot (*Perameles bougainville*) were successfully reintroduced into predator-free sections of the reserve between 1998 and 2001 (Moseby *et al.* 2011). This study was conducted within the Main Enclosure of the Reserve which was the first section to be completed (Fig. 1).

The climate is hot and dry with a long term average rainfall of 166 mm per annum. The mean annual summer maximum temperature exceeds 35° C, and the mean annual winter minimum is 4° C. The Reserve and surrounding area supports a variety of habitats including dunes (*Acacia ligulata* and *Dodonaea viscosa*), sandplains (*A. aneura*), chenopod swales (*Atriplex vesicaria* and *Maireana astrotricha*), ephemeral swamps (*Eragrostis australasica*), claypans and creeklines. Taller vegetation is present on dunes, whereas swales support more open, low vegetation (Finlayson and Moseby 2004). Feral cats, red foxes and European rabbits (*Oryctolagus cuniculus*) were present outside the Reserve, which is primarily used for cattle (*Bos taurus*) grazing.

#### **6.2.1.2 Greater Bilby**

The greater bilby is a medium-sized omnivorous marsupial that has declined significantly since European settlement. Although wild populations still occur in Western Australia, Northern Territory and Queensland, the species now occupies less than 20 % of its former range (Southgate 1995). The bilby is listed as vulnerable to extinction under the Australian Environmental Protection and Biodiversity Conservation Act (1999) with predation from introduced cats and foxes thought to be largely responsible for its decline. Male animals weigh up to 2.5 kg and females around 1 kg (Johnson 2008). Bilbies are nocturnal and largely solitary, using their highly developed sense of smell to locate food. Bilbies live in simple burrows and are proficient diggers, excavating foraging pits during their search for seeds, invertebrates and other food (Johnson 2008). Bilbies have a number of burrows within their home range and move between burrows over time (Moseby and O'Donnell 2003). Captive-bred bilbies were reintroduced into the main enclosure of the Arid Recovery Reserve in April 2000 (Moseby and O'Donnell 2003). Released bilbies were only provided with supplementary food and water for 3 months after release and so the population was considered wild at the time the experiment commenced five years later. The population was estimated at over 200 individuals when the experiment began in 2005 and due to the bilbies' short life span, only progeny of the released animals were captured during the experiment.

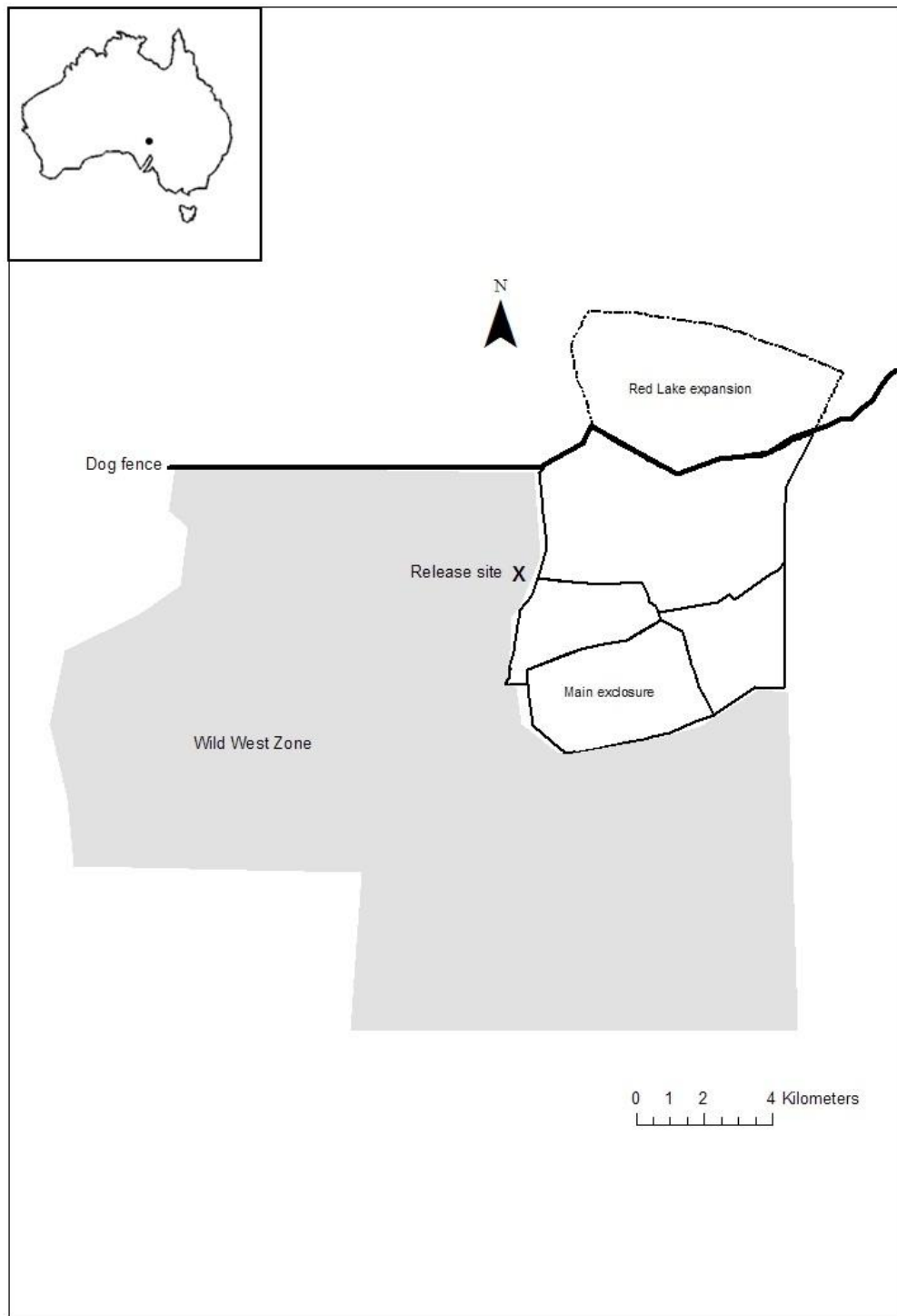


Figure 1: The Arid Recovery Reserve in northern South Australia showing the Main Exclosure inside the Reserve used during stage one and Wild West release site used during stage two.



### **6.2.1.3 Stimulus model and olfactory stimuli**

Bilbies were trained to associate an unpleasant experience with predator stimuli. Predator stimuli included model, tactile and olfactory stimuli. Model and tactile stimuli comprised an intact feral cat carcass that had been shot or trapped using Victor Soft-Catch™ (No. 1.5) soft-catch foot-hold traps (Coast to Coast Vermin Traps, Western Australia) outside the Reserve up to a month prior to the experiment. Traps were set during the day and checked each morning within two hours of dawn. All trapped cats were in traps for less than 24 hours. No injuries to captured cats were recorded when using soft-catch foot-hold traps and trapped cats were euthanased using a single .22 calibre rifle shot to the head. Seven cats were used during the study and were stored in a freezer below 0 ° C and defrosted the day before use. Both male and female carcasses were used and all cats had their claws removed prior to training. Olfactory stimuli comprised a mixture of cat urine and faecal matter extracted from feral cats that had been shot or trapped. Each faecal sample was placed in a 300 ml specimen jar and sufficient water added to cover it. The jar was shaken vigorously to break up the sample, and then put through a sieve to collect the liquid component. Minimal water was added to create a pungent liquid mixture of scent spray which could be administered through a 600 ml gardening spray bottle. Excess vials of the mixture were frozen until required and stored below 4°C.

### **6.2.1.4 Predator training**

During May 2005, a vehicle and handheld spotlight was used to locate bilbies along roads within the reserve at night. Seven bilbies were captured at random by two handlers with torches and handheld nylon fishing nets, but only those weighing more than 450g were used in the study. A third observer carried the cat carcass which was thrust on top of the bilby as soon as it was captured in the net. The cat was forcibly moved onto the bilby in an attempt to simulate a cat pouncing on prey. To avoid damage to the bilby, the cats' head and teeth were never placed in contact with the bilby. The bilby was then transferred to a fleece bag impregnated with scent spray. Scent spray was also sprayed on the cat carcass, nylon fishing nets and handler's gloves prior to and between each bilby capture.

Bilbies were then weighed on electronic scales and assessed for condition by feeling the fat stores between the hips and spine (classed as poor, fair, good or excellent). Pouch condition was assessed and bilbies were fitted with a Trovan ID100 microchip (11mm x 2.2mm, Microchips Australia) inserted under the skin between the shoulder blades. Microchips have been used for over 10 years on bilbies at Arid Recovery with no recorded incidence of microchip movement in recaptured animals. A 6-g radio-transmitter (<1.5% of body weight of the smallest bilby) from Biotelemetry Tracking (Adelaide) was attached dorsally to the base of their tail. Excess hair was removed from the bilbies tails with scissors and the remainder shaved with disposable razors, before attachment of the transmitters using Premium Leukoplast adhesive tape. Tail transmitters were successfully used on bilbies during their original release into the reserve in 2000 (Moseby and O'Donnell, 2003). Only experienced handlers attached tail transmitters as tail ulcers can develop if the tape is applied too tightly. If the bilby is not recaptured within 3-4 months, the tape loses its adhesive quality and slides off the tail. On release at the point of capture, trained bilbies were prompted out of the bag using the cat carcass. Two sprays of scent spray were aimed in the direction of the bilby as it fled from the release point.

Four of seven control bilbies were captured using the same netting method but not subject to the training treatment. They were processed and fitted with radiotransmitters in an identical

manner to trained bilbies and released at point of capture. The three additional control bilbies were opportunistically caught in wire cage traps placed around active burrows and baited with rolled oats and peanut butter. Cage traps were covered in hessian sacks to reduce stress and were checked at 9 pm and before dawn. Animals captured in cage traps received the same handling and processing procedures as other controls. To simulate capture in nets, these bilbies were pursued with nets after release until the animal had sought shelter down a burrow or eluded the researcher. Due to logistical constraints, seven control and seven trained bilbies were captured and trained in two sessions, three and a half weeks apart. During the first session, three control and three trained bilbies were captured over a six night period and during the second session four control and four trained bilbies were captured over 11 nights. Control animals were caught with clean equipment and on different nights to avoid contamination with urine spray on nets, handling bags and gloves. The exception was one night where a control animal was captured prior to two experimental animals. At the end of the study, animals were recaptured using pen traps, where a wire netting fence is erected around the diurnal burrow system and wire cage traps placed within holes cut in the netting (Southgate *et al.* 1995). Captured animals were weighed, checked for body condition, transmitters were removed and they were released at point of capture.

#### **6.2.1.5 Reinforcement**

In order to reinforce the capture training event, a second aversion event was paired with the scent spray stimuli six days after capture (Fig. 2). The aversion event aimed to mimic a mammalian predator attempting to dig up the bilby in its burrow. Trained bilbies were radiotracked to their diurnal burrow and the main entrance identified. The area around each entrance and the surrounding burrow was swept free of tracks. Three sprays of scent were administered down into the main entrance. Immediately after spraying, a short three-pronged gardening hoe was used to vigorously shift sand around the burrow entrance for three minutes. This resulted in a slight enlargement of the burrow entrance as sand was removed and raked back away from the opening. Excess sand was levelled around the burrow, before cat scent was resprayed down the excavated entrance. Control bilbies were subjected to the same reinforcement action but without the associated scent spray. During the first session, reinforcement burrow digging was undertaken three times with three days separating each burrow digging event. After it became apparent that session one bilbies became habituated to the reinforcement burrow digging (see results below), only one burrow digging event was conducted during the second session rendering this session slightly shorter in duration than the first session (29 vs 19 days, Fig. 2).

#### **6.2.1.6 Monitoring and testing**

Three days after capture but before the reinforcement burrow dig, a predator-avoidance training test was conducted on both control and trained bilbies. Bilbies were radiotracked to their diurnal burrows and the main burrow entrance identified as the largest entrance closest to the bilby transmitter or the one with the freshest and most numerous bilby tracks. All burrow entrances and sand around the burrow site were swept free of tracks. Three sprays of cat scent were administered onto the sandy floor of the main burrow entrance, 30 cm into the burrow to prevent obstruction by wind and or sand. All tests were undertaken within 4 hrs of sunset to ensure the scent was still fresh as bilbies left their burrows. Burrows were revisited the following day to record whether the bilby had moved away from the test burrow to

another diurnal burrow system. This test was repeated one week after the final reinforcement burrow dig occurred (Fig. 2).

Movements of trained and control bilbies were compared by radiotracking each bilby daily and locating their diurnal burrow. The GPS location of each diurnal burrow and the distance from the previous day's burrow was recorded. The number of burrow entrances and level of activity at each burrow was recorded. All entrances and the area within 1 m surrounding the burrow site were swept free of tracks so that new observations could be made the following day.

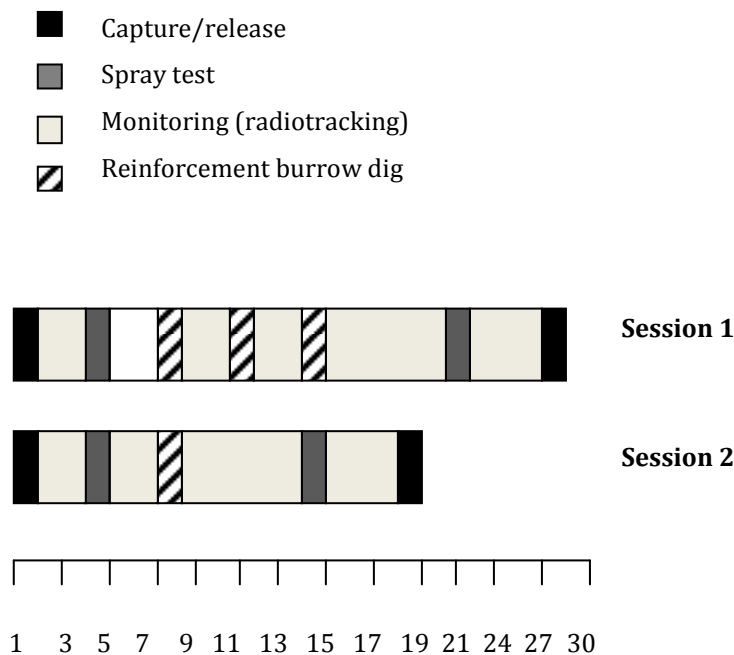


Figure 2: Training and testing schedule for bilbies caught in session one and session two. X axis is days of treatment.

### **6.2.1.7 Data analysis**

To assess the response of trained and control bilbies to the initial training, reinforcement burrow digging and spray tests, several factors were compared. The total number of different diurnal burrows used, the number of burrow entrances and the number of times each bilby changed burrows over the monitoring period (including returning to old burrows) were compared between trained and control bilbies using independent samples student's t-tests. The proportion of trained and control bilbies that moved burrows immediately after the spray test or reinforcement burrow dig event was compared using chi-squared analysis. In addition, the cumulative distance moved by trained and control bilbies between diurnal burrows over the monitoring period and the greatest linear distance between any two burrows used by each bilby were compared using independent samples student's t-tests. Levene's test was used to test for equality of variances, samples with unequal variances were transformed using  $\ln+1$ . We also compared the distance moved by each trained bilby the day after the initial spray test compared with its average daily movements over the eleven monitoring days. Bilby weight

and condition were compared between the initial capture and removal of the transmitters. Statistical analysis was undertaken using the computer software package SPSS Statistics 17.0. To determine if slight weight differences of bilbies between treatments influenced results, linear regressions were used to compare the weights of all bilbies with total distance moved, number of burrows used and the number of burrow entrances.

Due to the assumed habituation of bilbies to the digging events during session one, session two bilbies were only subjected to a single digging event. Thus in order to combine the burrow use and movement results from both sessions, statistical analysis was only conducted on data for the first 11 days, up until just before the second reinforcement dig. The response of the bilbies to the second spray test was analysed separately for session 1 and session 2 trained bilbies as the perceived habituation of the first session bilbies to the three burrow digging events was thought to affect their response to the final spray test.

## 6.2.2 STAGE TWO- PREDATORS PRESENT

### **6.2.2.1 Study site**

The Wild West Zone is a 200 square km unfenced release area outside the Arid Recovery Reserve, located immediately adjacent to the western fenceline of the Reserve and bordered to the north by the dog fence. The dog fence is a man-made wire netting fence designed to exclude dingos (*Canis lupus dingo*) from southern sheep grazing areas. Only the Arid Recovery fenceline forms a significant barrier to bilbies, cats and foxes. Rabbits, cats and foxes were all present in the Wild West Zone but ongoing control of cats and foxes limited their numbers. Cats and foxes were controlled during the study using a combination of bimonthly ground baiting with Eradicat™ (Department for Environment and Conservation, Western Australia) sausage baits poisoned with 1080 (density of 10-25 per km<sup>2</sup>), opportunistic poison baiting in areas where feral cat tracks were observed, weekly shooting and permanent trapping at up to 12 sites using Victor Soft-Catch™ (No. 1.5) soft-catch foothold traps (Coast to Coast Vermin Traps, Western Australia) using the same trapping method outlined previously. Habitats were similar to those within the Arid Recovery Reserve. The area is situated on the Roxby Downs Pastoral Station and was subjected to low levels of cattle grazing during the study. Numerous rabbit warrens were present in the release area and rabbit control was not attempted. Spotlight counts in the region estimated rabbit numbers at approximately 50 per square kilometre during the study (BHP Environmental Dept, Olympic Dam unpublished data).

### **6.2.2.2 Release of trained and untrained bilbies**

During August 2007, ten predator-trained and ten control bilbies were captured within the Arid Recovery Reserve and released outside into the Wild West Zone (Fig. 1). No supplementary food or water was provided. The release site was chosen in an extensive area of sand dunes, a habitat type preferred by bilbies (Moseby and O'Donnell 2003) and which contained an abundance of rabbit burrows for shelter. Capture and release occurred over three consecutive nights with all bilbies released into the Wild West Zone on the night of capture. Trained animals were captured in nets using the training method outlined previously. Control animals were captured in nets and burrow traps. Burrow traps were wire netting cage traps that were placed down active burrow entrances so that the bilby was captured emerging from

its burrow. Only one third of the trap protruded from the burrow ensuring the captured bilby had shelter from aerial predators and the weather. Traps were checked 3-4 hours after sunset and again before sunrise. Control and trained bilbies were captured on the same nights but different cars, nets, bags and gloves were used for capture and transport. All animals were processed at an enclosed shed within the Arid Recovery Reserve where a tail-mounted radio transmitter (Titley Electronics, Victoria) with a mortality trigger was attached dorsally to the base of their tail using the method outlined previously. The weight, sex, breeding status and condition were recorded for each individual.

### **6.2.2.3 Monitoring**

Attempts were made to locate all released bilbies daily for the first month after release followed by weekly mortality checks. An observer on quad bike was used to locate animals with a hand held receiver and antenna, occasionally a Cessna light aeroplane was used to locate bilbies for mortality checks. The burrow location, number of entrances and spoor of predators and co-habiting species were recorded. Burrows were also scored as being existing rabbit warrens or new burrows that had likely been excavated by bilbies. The mortality rate, total distance between burrows and the number of burrows used was compared between trained and control bilbies. Comparison of the number of burrows used and distance between burrows were conducted at 30 days and also at 11 days after release to enable direct comparison with the bilby training events inside the Reserve. After 2-3 months, any surviving bilbies were recaptured using pen and burrow traps and their transmitters replaced. Transmitters fell off or were removed after six months. Predator presence on regional spoor transects was recorded during the study as per methods outlined in Moseby *et al.* (2011). After four months, when results suggested high survival of both trained and control bilbies, a spray test was conducted at burrows occupied by both trained and control bilbies to determine the proportion of bilbies that moved burrows.

## **6.3 RESULTS**

### ***6.3.1 STAGE ONE- PREDATOR FREE ENVIRONMENT***

A total of seven trained (4M, 3F) and seven control bilbies (4M, 3F) were captured and used in the study (Table 1). An additional male control bilby was also captured but its erratic behaviour, much larger body weight and extremely large home range size made it difficult to monitor using radiotracking and suggested it should be treated as an outlier and excluded from the experiment. The weights and body condition of trained and control bilbies were similar (Fig. 3). Although the average weight of trained bilbies was slightly heavier than control animals (trained 440-1507 g mean=878 g  $\pm$  SE=128; control 519-875 g mean 673 g  $\pm$  SE= 51, Table 1), there was no significant difference between the two treatments ( $t=1.48$ ,  $df=1$ ,  $P=0.166$ ). All bilbies were scored as having either good or excellent body condition at the time of capture. There was some weight change recorded between the start and end of the study with a similar number of trained and control bilbies either losing or gaining weight (Table 1). One trained bilby died in its burrow when a pen trap was set to recapture it and remove its transmitter.

Trained bilbies changed burrows significantly more often than control bilbies ( $t = 4.531$ ,  $d.f. = 12$ ,  $P = 0.001$ ). Over the first eleven monitoring days, trained bilbies changed burrows on average 5.7 times, compared to control bilbies that averaged 1.42 burrow moves. As a result, trained bilbies also used significantly more burrows (average 3.85) compared to control bilbies (2.14) ( $t = 3.2$ ,  $d.f. = 12$ ,  $P = 0.007$ ). There was a significant difference in the number of entrances per burrow used by trained and control bilbies ( $t = -2.184$ ,  $d.f. = 38$ ,  $P = 0.035$ ) with trained bilbies using burrows with 1-8 entrances (average 3.6) and control bilbies using burrows with 1-5 (average 2.2) entrances (Fig. 4).

The average total distance moved between successive diurnal burrows by trained bilbies over the 11 day monitoring period was 1387 m ( $\pm 513$  m), considerably higher than the average of 158 m ( $\pm 40$  m) moved by control bilbies ( $t = 2.387$ ,  $d.f. = 12$ ,  $P = 0.034$ ). The distance between the two furthest diurnal burrows of each bilby revealed a weak but not significant difference between trained (average = 352 m) and control (average = 147 m) bilbies ( $t = -2.104$ ,  $d.f. = 12$ ,  $p = 0.057$ ).

There was no indication that the slight but not significant weight differences between trained and control bilbies influenced results. Linear regressions comparing the relationship between bilby weight and total distance moved ( $R^2 = 0.004$ ,  $F = 0.044$ ,  $df = 1, 13$ ,  $P = 0.838$ ) average number of entrances per burrow ( $R^2 = 0.059$ ,  $F = 0.746$ ,  $df = 1, 13$ ,  $P = 0.405$ ) and number of burrows used ( $R^2 = 0.004$ ,  $F = 0.043$ ,  $df = 1, 13$ ,  $P = 0.839$ ) were not significant.

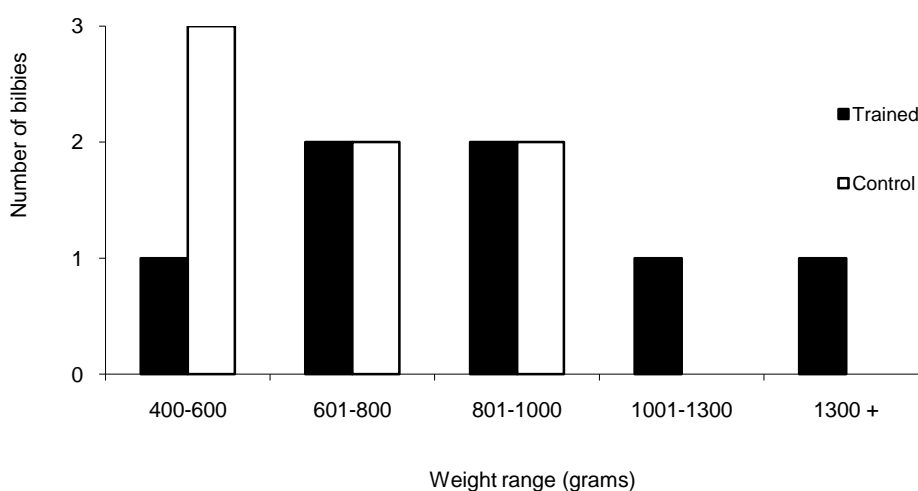


Figure 3: Weight range of control and trained bilbies used during stage one (predator free environment).

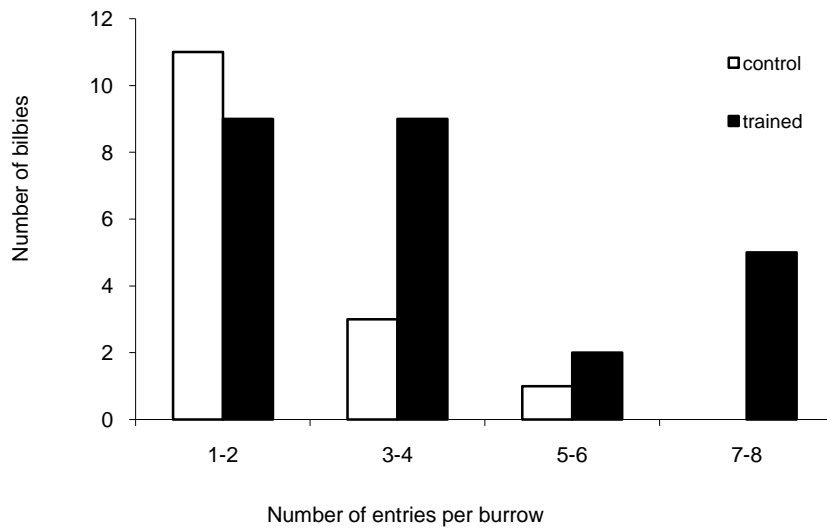


Figure 4: Frequency of burrows with different number of entry points used by control and trained bilbies during stage one (predator free environment)

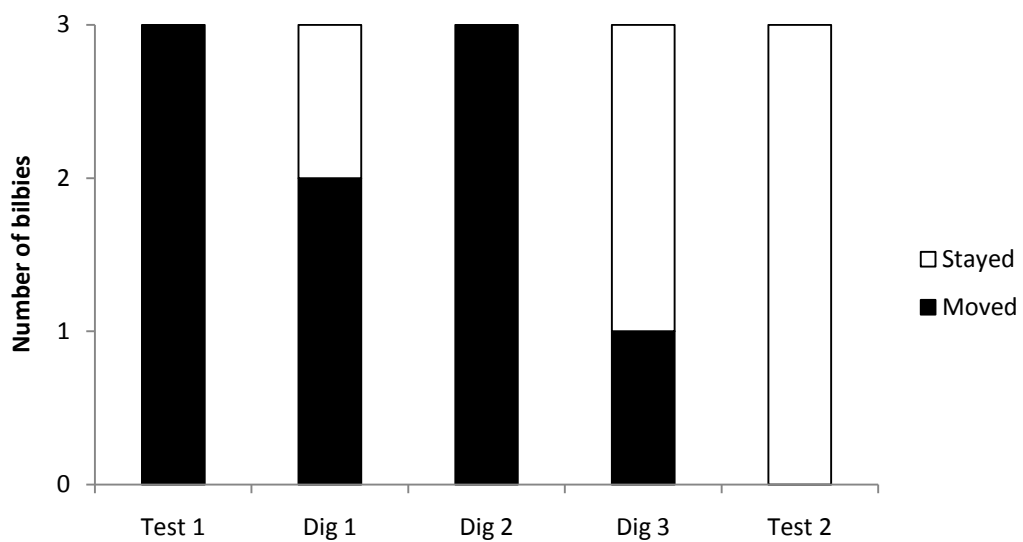


Figure 5: Burrow movement of session one trained bilbies indicating that they may have become acclimatised to the burrow dig reinforcement events.

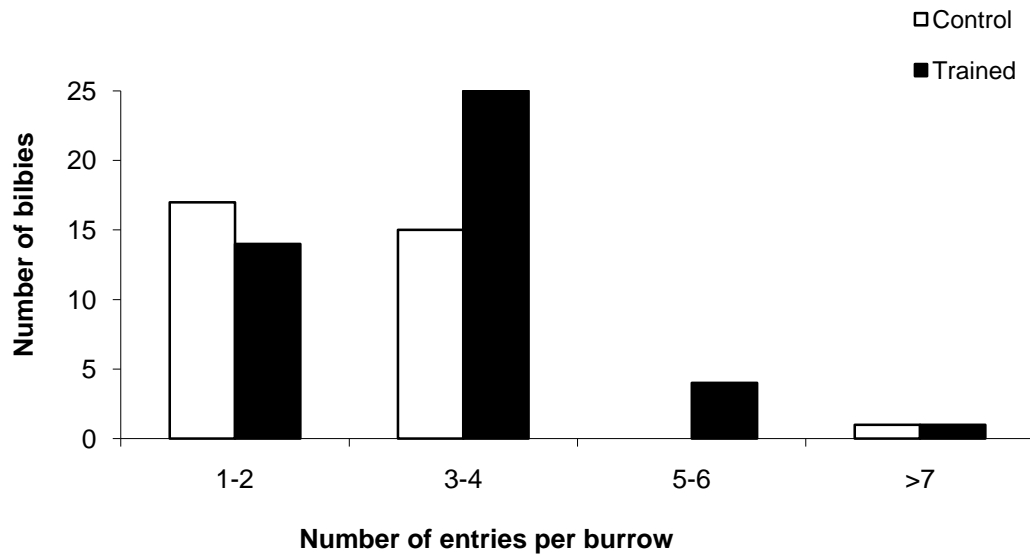


Figure 6: Frequency of burrows with different number of entry points used by control and trained bilbies during stage two (outside the Reserve where predators were present).



Table 1: Details of the bilbies released during stage 1 inside the Reserve where cats and foxes were absent. Y=moved burrows after entrance was sprayed with cat urine. N=remained in same burrow. Missing data in test 2 is session 1 bilbies.\*=bilby died in burrow and was not reweighed.

Sex	Session	weight release	weight 1 month	Weight difference	condition release	condition 1 month	Test 1	Dig 1	Test 2	No burrows used	No times moved	Total dist moved
<i>Control</i>												
M	1	578	622	44	Good	Good	N	N	-	1	0	0
M	1	519	587	68	Good	Excellent	N	N	-	2	1	220
M	2	680	707	27	Good	Good	N	N	N	3	4	270
M	2	551	570	19	Excellent	Excellent	N	N	N	3	2	110
F	1	875	867	-8	Excellent	Fair	N	N	-	2	1	70
F	2	826	819	-7	Excellent	Excellent	N	N	N	2	1	150
F	2	685	637	-48	Excellent	Excellent	N	N	N	2	1	290
<i>Trained</i>												
M	1	440	433	-7	Excellent	Fair	Y	N	-	4	4	1270
M	1	653	668	15	Good	Good	Y	Y	-	6	9	4270
M	2	1507	1292	-215	Excellent	Fair	Y	N	Y	3	3	470
M	2	806	850	44	Excellent	Excellent	Y	Y	N	2	3	150
F*	2	1062	-	-	Excellent	Died	Y	Y	Y	4	8	1390
F	1	930	993	63	Good	Excellent	Y	Y	-	4	7	790
F	2	750	700	-50	Excellent	Excellent	Y	Y	Y	4	6	1370

Table 2: Details of the bilbies released during stage 2 outside the Reserve where cats and foxes were present. Despite significant effort, not all bilbies could be recaptured after 3 months. Y=moved burrows after entrance was sprayed with cat urine. N=remained in same burrow. ^= natural causes \*=predation or scavenging by Wedge-tailed Eagle

Bilby No.	Sex	Condition capture	Date captured	Method	Weight release	Weight 3 months	Condition 3 months	Test Spray	Deaths
<i>Control</i>									
C1	M	Fair	21-Aug-07	Net	643	1118	Good		
C2	M	Good	21-Aug-07	Trap	756				
C3	M	Good	21-Aug-07	Net	1002	1222	Excellent		
C4	M	Poor	21-Aug-07	Trap	750				
C5	F	Good	21-Aug-07	Trap	816				Died^-19 days
C6	F	Fair	22-Aug-07	Trap	948	1186	Good	Y	
C7	F	Fair	22-Aug-07	Trap	969	1181	Excellent	Y	
C8	F	Good	23-Aug-07	Net	808	1030	Excellent	Y	
C9	F	Good	23-Aug-07	Net	713	838	Excellent	N	
C10	F	Fair	23-Aug-07	Trap	726	894	Good	N	
<i>Trained</i>									
T1	F	Good	21-Aug-07	Net	619	947	Excellent	N	
T2	F	Fair	21-Aug-07	Net	628			N	
T3	F	Fair	21-Aug-07	Net	728	1130	Good	Y	
T4	M	Fair	22-Aug-07	Net	1045				
T5	F	Good	22-Aug-07	Net	962	986	Excellent	Y	
T6	F	Excellent	22-Aug-07	Net	901	1056	Excellent		
T7	F	Good	22-Aug-07	Net	527				Died*- 27 days
T8	F	Fair	22-Aug-07	Net	720	962	Excellent	Y	
T9	M	Fair	22-Aug-07	Net	611	886	Good		
T10	M	Fair	23-Aug-07	Net	1151				

### **6.3.1.1 Reinforcement and testing**

All seven trained bilbies moved burrows after the initial spray test whilst all seven control bilbies remained in the same diurnal burrow. Track observations of control bilby burrows identified that all untrained animals had left their burrows after the test but returned the same night. Interestingly, only three of the seven trained bilbies returned to the sprayed burrow during the study and only after a minimum of three days, by which time the feline scent was not detectable to humans. Trained bilbies moved an average of 250 m from the spray test burrow after the test compared with average movements of 200 m between diurnal burrows at other times.

When the results of the first burrow reinforcement dig were compared between trained and control bilbies, trained bilbies were significantly more likely to move burrows (5 out of 7 moved), than control bilbies which all remained at the burrow where the reinforcement dig had occurred ( $\chi^2 = 7.77$ , d. f. = 1,  $p < 0.01$ ).

The results of the second spray test were analysed separately for session 1 and session 2 bilbies due to the higher number of burrow dig reinforcements in session 1 animals. During the second spray test on session two animals, all four control bilbies again remained at the same burrow while three out of the four trained bilbies moved burrows. Session one animals were subjected to three reinforcement burrow digs and the number of trained bilbies moving burrows after burrow digs and spray tests declined over time (Fig. 5). During the second spray test, no trained bilbies moved burrows whilst one of the three control animals moved. Low sample sizes prevented statistical analysis.

### **6.3.1.2 Behaviour**

During weighing and handling, trained bilbies were generally more agitated compared with control bilbies during both initial capture and final capture when a clean bag was used. Whilst control bilbies sat quietly in the bag, most trained bilbies struggled during handling, scabbling at the bag with their claws and trying to escape. Additionally, when trying to recapture bilbies to remove their radio transmitters, trained bilbies were much harder to trap than control animals. Six out of seven control bilbies were captured in pen traps on the first night compared with trained bilbies that required up to five nights of trapping. One trained bilby had a pen trap erected but did not emerge from its burrow and died down the burrow on the third night of trapping. Future trapping protocols were changed to ensure only two consecutive nights of trapping were conducted before traps were removed and trapping resumed at a later date. Food was also placed at the burrow entrance on the second night of trapping.

## ***6.3.2 STAGE TWO- PREDATORS PRESENT***

Ten control (4M, 6F) and ten trained (3M, 7F) bilbies (Table 2) were released into the Wild West Zone in August 2007. Seventeen of the 20 released bilbies were still alive six months after release with only one death attributed to a ground predator, a control bilby killed by a cat four months after release. One male trained bilby was killed or scavenged by a wedge-tailed eagle 27 days after release. Its remains were located under an eagle roost that contained numerous regurgitated pellets, bird droppings and other carcasses. A control female bilby

died in her burrow 19 days after release. Post mortem analysis by Zoos S.A. could find no evidence for a cause of death and the death was attributed to natural causes. No female bilbies carried pouch young at release but all females recaptured after three months were carrying pouch young measuring 60-70 mm from crown to rump. All recaptured bilbies had increased in weight and condition (Table 2).

Regional cat and fox activity on track transects during the first six months after release averaged 20-30% and 10% respectively (Moseby *et al.* 2011). A scent spray test was conducted 4 months after release on five trained and five control bilbies. Only female bilbies could be located on the day of the spray test and results were identical for both trained and control groups (Table 2). Two animals in each group remained in their spray burrow and three bilbies moved burrows. Of the three trained and control bilbies that moved burrows, only two trained bilbies had moved more than 10 m from the test burrow.

Two to three months after release we attempted to recapture 15 bilbies. Thirteen bilbies were recaptured, two trained bilbies could not be recaptured despite numerous attempts. Only two bilbies (trained) were caught on the first night of trapping and the remaining bilbies were difficult to trap and required between 2 and 7 trap nights (average=3).. Radio transmitters fell off or were removed after 6 months and track counts indicated that bilbies gradually declined in the release area until they became extinct 19 months after release (Moseby *et al.* 2011). Juvenile bilby tracks were observed after pouch exit at several independent burrows but disappeared soon after cat tracks were observed at all their burrow entrances.

Due to logistical difficulties and the large number of bilbies being monitored, only a proportion of the released bilbies could be radiotracked to burrows each day. The average number of diurnal burrow fixes collected over the first 30 days of monitoring was 11.9 for control animals and 14 for trained bilbies. In order to standardise the data and enable comparisons between trained and control bilbies, the number of days that fixes were obtained for each bilby was divided by the total number of burrows it used over that period. The cumulative distance between successive burrows was also divided by the number of radiotracking days. There was no significant difference in the cumulative distance moved between burrows for trained and control bilbies during the first 11 or 30 days after release. Control bilbies averaged 235 m and trained bilbies 172 m between diurnal burrow fixes over the 30 days. There was also no significant difference in the number of burrows used over either the first 11 or 30 days with 2.1 fixes per burrow recorded for control bilbies and 3 for trained animals over the 30 day period. The bilby that died after 19 days was excluded from the 30 day analysis but included in the 11 day analysis. The majority of burrows used by bilbies in the first 30 days after release were existing rabbit warrens (trained- 93% , control- 97%) and only five burrows were dug by bilbies over this period. At 11 days post-release there was no significant difference in the number of burrow entrances at burrows used by trained and control bilbies (Fig. 6). This trend was also apparent at 30 days post release.

## 6.4 DISCUSSION

### 6.4.1 STAGE ONE- PREDATOR FREE ENVIRONMENT

This study provides the first evidence that wild naive bilbies can be trained to respond to predator scent by associating their capture with an unpleasant experience. Significant differences in movement patterns and burrow selection were recorded between trained and control bilbies suggesting that the model and tactile stimuli and/or scent spray triggered a behavioural response.

When animals encounter a recognised predator they possess innate defence reactions which can include fleeing, freezing or adopting some kind of threat behaviour (Bolles 1970). The movement of trained bilbies away from each burrow site where feline scent had been sprayed suggests the trained bilbies were exhibiting antipredator behaviour by fleeing and trying to avoid an unpleasant experience or potential predator. Control bilbies did not move burrows suggesting that they did not recognize the scent of a potential predator or did not perceive it to be a threat. It could also be argued that the control bilbies responded to the olfactory cue by not moving burrows, however observations of control bilby burrows showed that all control animals left the test burrow at some point in the night to forage but then later returned. Our results support other studies that have shown prey species will change their behaviour in response to predator odours (Sullivan and Crump 1984; Kats and Dill 1998; Perot-Sinal *et al.* 1999) and that only predator-exposed animals will react to olfactory cues (Blumstein *et al.* 2002).

Trained bilbies also moved a greater total cumulative distance over the monitoring period and used more burrows compared to the control animals, suggesting increased movement by trained bilbies. Trained bilbies may have been continually moving to avoid detection from predators or trying to move away from the vicinity of the potential predator. Although some studies have shown that travelling further distances can actually increase the risk of encountering a predator (Steen 1994), other studies have shown that remaining in one place can also increase an animal's predation risk, particularly for species that accumulate scent and faecal deposits (Banks *et al.* 2002). Predators such as foxes and dingoes will locate and scent mark warrens of prey species and return to them to prey on resident animals. Continually moving burrows and leaving areas where predators are present is likely to be a successful predator avoidance strategy for a wide-ranging fossorial species avoiding ground-based predators.

Trained bilbies were also more likely to use burrows with more entrances. Ground predators such as dingoes, foxes and monitor lizards will dig up burrows to prey on resident rabbits and a burrow with multiple entrances provides more escape opportunities compared with a single-entrance burrow. A study by Hirsch and Bolles (1980) compared ground squirrel behaviour when faced with model aerial and ground predators. When faced with an aerial model, squirrels escaped to a burrow regardless of the number of entrances, but when faced with a ground predator model the squirrels always escaped to a multi-entranced burrow. Another advantage of using a multi-entranced burrow is that they are more likely to house other conspecifics or animals of a different species (Read *et al.* 2008) thereby increasing the chance of an individual surviving if the burrow is attacked by a predator. In this study, bilbies were

often found to share burrows with conspecifics and burrowing bettongs and in other areas, bilbies have been recorded sharing with rabbits (P. McRae pers. obs.).

The combined simultaneous use of model, tactile and olfactory cues may have helped elicit such a strong behavioural response in trained animals. Bouskila & Blumstein (1992) argue that animals using olfactory cues alone are likely to make errors in estimating predation risk and Bradbury and Vehrencamp (1998) suggest that olfactory cues may persist long after the predator has moved away thus not providing information about the probability of predation. Inert model stimuli are also unlikely to represent a real life predator situation. Some training studies use aversion events such as water pistols (McLean *et al.* 1996), rubber bands (Miller *et al.* 1990) and loud noises during training but we feel that the aversion event and stimuli used in our study were more realistic and likely to trigger an appropriate response. Chasing a bilby during its normal foraging activity and placing a cat carcass over it when captured closely mimics a real predation attempt in the wild. By pairing the visual model stimuli with tactile and olfactory stimuli, three sensory pathways are triggered, potentially leading to more vivid memory retention. This is supported by the fact that trained bilbies were highly agitated during recapture and handling and extremely difficult to recapture with one bilby seemingly dying in its burrow rather than emerging and being recaptured.

The pairing of a single aversion event with a visual, tactile and olfactory cue was sufficient to change the behaviour of trained bilbies. Many predator avoidance studies using model predators have exposed the treatment animals to visual stimuli up to six times before conducting post training tests (see Blumstein *et al.* 2000). McLean *et al.* (2000) presented wallabies with fox models four times with aversive training before conducting post training tests. From many of those studies however, learning occurred after only one or two training trials and prolonged training events led to habituation (Griffin *et al.* 2000). The result of this study confirms that multiple events are probably undesirable as the three reinforcement digs used during session one animals triggered habituation and weakened the anti-predator response. Interestingly, all the control bilbies from the first session remained at the same burrow after each of the three burrow digging events, highlighting that the control bilbies either did not associate the burrow digging with a predatory encounter or did not feel threatened enough to move burrows.

Responses of treatment bilbies to the anti-predator training were shown to persist up until three days after the final spray test, approximately 19 days after initial capture. In some experimental studies, animals have been shown to retain responses to learned stimuli for 30 days (Miller *et al.* 1990), two to three months (Mineka and Cook 1988) and up to 8 months (McLean *et al.* 1996). Our trained animals were not released into a predator environment and it is not known how long the training would last in the absence of predators.

The health of both trained and control bilbies was largely unaffected by the project and suggests that the training did not negatively affect the animals and that only their behaviour was modified. Bilby weight appeared to have no effect on their response to the training or tests, which also suggests that bilbies varying in size and weight can potentially be trained in predator avoidance behaviour. However, further studies would be required on juvenile (<450 g) animals to determine their response. The three control bilbies which were captured opportunistically using cage traps appeared to respond similarly to the other control bilbies which were caught using nets suggesting that further studies may incorporate the use of both catching techniques. The main constraint with this study was the inability to gather baseline data on the bilbies prior to the initial training event. Thus pre and post training behaviour could not be compared for individual animals. However, this would potentially prove quite

difficult to undertake, particularly trying to find and net pre-selected bilbies after they had been previously captured.

Although we used visual stimuli during the initial training event, we did not test to see whether trained and control bilbies responded differently to the sight of the cat after training. In future studies it may be worthwhile to monitor the response of trained and control animals by placing a model cat outside their burrows and recording their behaviour with remote cameras. It would also be useful to test other predator models and scent spray to determine if trained bilbies are able to generalise their response to other predators. Predator archetypes are thought to elicit a similar response in some prey species (Cox and Lima 2006) and all carnivores produce similar sulfurous metabolites from meat digestion suggesting that prey may generalize faecal deposits from different predators (Nolte *et al.* 1994). Griffin *et al.* (2002) found that tammar wallabies trained to fear a model fox generalised their fear to a model cat which was not paired with the aversion event. In this study the decision to use a cat carcass and feline scent over that of a fox (an equally effective predator of bilbies) was based on availability and because cats in the region are more difficult to control (Moseby and Hill 2011) and caused previous reintroduction failure in the bilby (Moseby *et al.* 2011).

#### 6.4.2 STAGE TWO- PREDATORS PRESENT

The only bilby to be killed by a mammalian predator during the first six months after release into the Wild West Zone was a control bilby but mortality rates were too low to determine whether training improved survival of released bilbies. In contrast to trials inside the Reserve, no differences in burrow use or movement patterns were found between trained and control bilbies during the first 11 or 30 days after release. Results from the spray test also suggest little change in behaviour between the two treatments. This discrepancy could be due to several possibilities including; the training did not change bilby behaviour, both trained and control bilbies changed their behaviour or trained bilbies exhibited different behaviour to control animals but we were not able to detect the behavioural change.

A failure to detect behavioural change is a possibility, particularly due to the low number of consecutive fixes obtained after release and the low level of mortality recorded. However, several factors suggest that in fact both trained and control bilbies may have changed their behaviour and become more aware of predators. Regional cat activity was similar during this study to the initial untrained 2004 release (both detected on 20-30 % of transects) where 8 of the 12 bilbies died from cat predation in the first month after release. The much higher survival rate of both trained and control bilbies in the present study suggests that both release groups were better able to avoid predators. The numerous trapping attempts required to recapture both trained and control bilbies after three months was similar to that required for trained animals inside the reserve, further supporting the suggestion that both release groups were more predator-aware. Avoidance behaviour of the trained group may have been acquired by the control group after release through cultural transmission (Griffin *et al.* 2000) as both groups were released into the same area and remained in close proximity after release. Bilbies, although mostly solitary, use scent marking to provide olfactory cues to conspecifics (Johnson and Johnson 1983) and have a prominent white tip on their tail which may be used to send visual cues. Other studies have shown that birds and monkeys can culturally acquire anti-predator behaviour from observing conspecifics (Cureo 1988; Mineka and Cook 1988; McLean *et al.* 1999). Both control and trained bilbies preferred to use available rabbit

warrens, many of which were occupied, rather than excavate their own warrens. Anti-predator behaviour of rabbits may also have been observed and learnt by co-habiting bilbies and/or predation risk may have been lowered by burrow sharing.

Additionally, control animals may have been exposed to some of the urine spray during capture and release, as control animals were captured on the same nights as trained animals. Despite keeping bags, gloves, nets and cars separate for the control and trained bilbies, traces of urine may have been present during handling as both groups were processed in the same location. Bilbies have an acute sense of smell which they use to locate food and mates and even traces of odours may have been enough to stimulate a change in behaviour. Both trained and control animals exhibited a similar response to the spray test at four months after release but this test was not conducted soon enough after release to be conclusive.

Finally, the presence of cats and foxes in the release area outside the Reserve coupled with the relocation of bilbies to a new environment may have been enough to stimulate some level of predator-avoidance behaviour without the need for training. If bilbies were displaying type 1 (failure to recognise predator) ontogenic naivety as suggested previously then natural exposure to predator odour after release may have prompted “self training”. The suggestion that animals can behave differently in laboratory and field situations is supported by findings by Fey *et al.* (2010) who found that predator naive voles did not avoid alien mink odours in the lab but did in the field. However, “self training” alone does not explain the higher survival rate in this release compared with the untrained release of bilbies outside the Reserve in 2004 and differences in food availability, rabbit numbers and/or predator numbers in the release area may have also contributed.

The behavioural changes in the trained bilbies inside the reserve suggest that they can be trained to exhibit appropriate predator-avoidance behaviour for ground predators. Results indicate that bilbies reintroduced from captive-bred sources exhibit type 1 (failure to recognise predators) ontogenic predator naivety as untrained animals did not appear to recognise the scent spray but trained bilbies implemented appropriate anti-predator strategies. However, several factors may limit the usefulness of predator-avoidance training in bilbies released into areas with predators. First, bilbies may ‘self-train’ when in unfamiliar environment where predators are present. Second, even if training increases predator awareness, evolutionary isolation may mean bilbies also exhibit type 3 predator naivety (*sensu* Banks and Dickman 2007), where they recognise an animal as a predator, initiate appropriate responses but a lack of co-evolution with the predator means they were out matched by the superior hunting skills. Thus, predator training may help reintroduced bilbies avoid native predators like dingoes but may be ineffective when smaller exotic predators such as cats and foxes, which are more likely to be able to hunt them in burrows, are present. Additionally, predator avoidance training is likely to be most effective in animals with altricial young that stay with the adult long enough for the training to be learned. Bilbies have an 80 day pouch life (Johnson 2008) and become independent soon after leaving the pouch when they can weigh as little as 170 g (Arid Recovery unpublished data; Southgate *et al.* 2000). Young independent bilbies are precocious, known to travel large distances and exhibit tame-like behaviour, they are relatively easy to catch and wander around in the open (K. Moseby pers. obs.). Results from the release outside the Reserve suggest that juveniles were surviving to pouch exit but succumbing to cat predation soon afterwards. These findings indicate that continued predator control is required to ensure successful recruitment and population growth.



To determine the effectiveness of predator-avoidance training, we recommend further investigation of cultural and filial transfer. To determine if learned behaviour is culturally transferred to conspecifics we suggest implementing separate, replicated release sites for control and trained bilbies within a region with uniform predator densities. Control and trained bilbies should be captured on different nights, similar to our initial trials within the Reserve. To investigate filial transfer we suggest comparing predator avoidance behaviour of juveniles recently weaned from trained and untrained mothers. Although initial results suggest bilbies can be trained to respond to predator cues, further work is needed to determine if this translates into a net benefit in the wild.

## 6.5 ACKNOWLEDGEMENTS

Funding for this study was provided by Arid Recovery, a conservation organisation supported by BHP Billiton, The South Australian Department of Environment and Natural Resources, The University of Adelaide and the local community. Supporting funds were obtained from the Natural Heritage Trust Envirofund, an Australian Federal Government grants program. We are indebted to the numerous staff and volunteers who assisted with this project including John Read, Jenny Newport, Chris McGoldrick, Brydie Hill, Adam Bester, Justine Smith and Peter Paisley. Phil Goldsworthy and Jim Spring assisted with flights and Zoos S.A. provided post mortem analysis. John Read kindly provided useful comments on the manuscript. This study was conducted under permits from the South Australian Wildlife Ethics Committee (4/2005 and 44/2005-M1).

## 6.6 REFERENCES

- Banks, P.B. & Dickman, C.R.** 2007. Alien predation and the effects of multiple levels of prey naivete'. *Trends Ecol. Evol.*, **22**, 229–230
- Banks, P.B.** 1998. Responses of Australian bush rats, *Rattus fuscipes*, to the odor of introduced *Vulpes vulpes*. *Journal of Mammalogy*, **79**, 1260–1264
- Banks, P.B., Norrdahl, K. & Korpimaki, E.** 2002. Mobility decisions and the predation risks of reintroduction. *Biological Conservation* **103**, 133-138.
- Blumstein, D.T., Mari, M., Daniel, J.C., Ardron, J.G., Griffin, A.S., & Evans, C.S.** 2002. Olfactory predator recognition: wallabies may have to learn to be wary. *Animal Conservation* **5**, 87-93
- Bolles, R.C.** 1970. Species-specific defense reactions and avoidance learning. *Psychological Review* **77**, 32-48
- Bouskila, A. & Blumstein, D.T.** 1992. Rules of thumb for predation hazard assessment: predictions from a dynamic model. *American Naturalist* **139**, 161-176.
- Bradbury, J.W. & Vehrencamp, S.L.** 1998. *Principles of animal communication*. Sunderland: MA, Sinauer.

- Cox, J.G. & Lima, S.L.** 2006. Naivete´ and an aquatic–terrestrial dichotomy in the effects of introduced predators. *Trends in Ecology and Evolution* **21**, 674–680.
- Curio, E.** 1988. Cultural transmission or enemy by birds. In: *Social learning: psychological and biological perspectives*. (T.R. Zentall and B.G. Galef Jnr) pp. 75-97. Lawrence Erlbaum Associates: Hillsdale, New Jersey.
- Fey, K., Banks, P.B., Ylönen C, H. and Korpimäki, E.** 2010. Behavioural responses of voles to simulated risk of predation by a native and an alien mustelid: an odour manipulation experiment. *Wildlife Research* **37**, 273-282.
- Finlayson, G.R., & Moseby, K.E.** 2004. Managing confined populations: The influence of density on the home range and habitat use of re-introduced burrowing bettongs (*Bettongia lesueur*). *Wildlife Research*, **31**, 457-463.
- Fischer, J., & Lindenmayer, D.B.** (2000). An assessment of the published results of animal relocations. *Biological Conservation*, **96**, 1-11.
- Griffin, A.S., Blumstein, D.T., & Evans, C.S.** 2000. Training captive-bred or translocated animals to avoid predators. *Conservation Biology*, **14**, 1317-1326.
- Griffin, A.S., Evans, C.S., & Blumstein, D.T.** 2002. Selective Learning in a Marsupial. *Ethology*, **108**, 1103-1114.
- Hirsch, S.M & Bolles, R.C.** 1980. On the Ability of Prey to Recognise Predators. *Z. Tierpsychol*, **54**, 71-84.
- Johnson, C. N. & Johnson, K. A.** 1983. Behaviour of the Bilby, *Macrotis lagotis* (Reid), (Marsupialia: Thylacomyidae) in captivity. *Australian Wildlife Research*, **10**, 77-87.
- Johnson, K.A.** 2008. Bilby. In: *Mammals of Australia*. (D. Van Dyck and R. Strahan) pp 191-193. Reed New Holland: Sydney, Australia.
- Kats, L. B., & Dill, L. M.** 1998. The scent of death: chemosensory assessment of predation risk by prey animals. *Ecoscience*, **5**, 361–394.
- Magurran, A.E.** 1989. Acquired Recognition of Predator Odor in the European Minnow (*Phoxinus phoxinus*) *Ethology*, **82**, 216-223.
- McLean, I.G., Lundie-Jenkins, & Jarman, P.J.** 1996. Teaching an endangered mammal to recognise predators. *Biological Conservation*, **75**, 51-62.
- McLean, I.G., Holzer, C., Studholme, B.J.S.** 1999. Teaching predator-recognition to a naïve bird. Implications for management. *Biological Conservation*, **87**, 123-130.
- McLean, I.G., Schmitt, N.T., Jarman, P.J., Duncan, C., & Wynne, C.D.L.** 2000. Learning for life: Training marsupials to recognize introduced predators. *Animal Behaviour*, **137**, 1361-1376.
- Miller, B., Biggins, D., Wemmer, C., Powell, R., Calvo, L., Hanebury, L & Wharton, T.** 1990. Development of Survival Skills in Captive- Raised Siberian Polecats (*Mustela eversmanni*) 11: Predator Avoidance. *Journal of Ethology*, **8**, 95-104.

- Mineka, S. & Cook, M.** 1988. Social learning and the acquisition of snake fear in monkeys. In: *Social learning: psychological and biological perspectives*. (T.R. Zentall and B.G. Galef Jnr) pp. 51-73. Lawrence Erlbaum Associates: Hillsdale, New Jersey.
- Moseby, K.E. & O'Donnell, E.O.** 2003. Reintroduction of the greater bilby, *Macrotis lagotis* (Reid) (Marsupialia: Thylacomyidae), to northern South Australia: survival, ecology and notes on reintroduction protocols. *Wildlife Research*, **30**, 15-27.
- Moseby, K.E. & Hill, B.M.** 2011. The use of poison baits to control feral cats and red foxes in arid South Australia 1. Aerial Baiting Trials. *Wildlife Research*, **38**, 338-349.
- Moseby, K.E., Read, J.L., Paton, D.C., Copley, P., Hill, B.M. & Crisp, H.M.** 2011. Predation determines the outcome of 11 reintroduction attempts in arid Australia. *Biological Conservation*, DOI10.1016/j.biocon.2011.08.003.
- Nolte, D.L., Mason, J.R., Epple, G., Aronov, E. & Campbell, D.L.** 1994. Why are predator urines aversive to prey? *J. Chem Ecology*, **20**, 1505-1516.
- Perot-Sinal, T. S., Ossenkopp, K. P. & Kavaliers, M.** 1999. Brief predator odour exposure activates the HPA axis independent of locomotor changes. *Neuroethology*, **10**, 775-780.
- Read, J.L., Carter, J., Moseby, K.M. and Greenville, A.** 2008. Ecological roles of rabbit, bettong and bilby warrens in arid Australia. *Journal of Arid Environments*, **72**, 2124-2130
- Short, J.** 2009. The characteristics and success of vertebrate translocations within Australia. Australian Government Department of Agriculture, Fisheries and Forestry: Canberra, Australia.
- Short, J., Turner, B.** 2000. Reintroduction of the burrowing bettong *Bettongia lesueur* (Marsupialia: Potoroidae) to mainland Australia. *Biological Conservation*, **96**, 185-196.
- Southgate, R., McRae, P. & Atherton, R.** 1995. Trapping techniques and a pen design for the Greater Bilby, *Macrotis lagotis*. *Australian Mammalogy*, **18**, 101-104.
- Southgate R.I., Christie P. and Bellchambers K.** 2000. Breeding biology of captive, reintroduced and wild greater bilbies, *Macrotis lagotis* (Marsupialia: Peramilidae). *Wildlife Research*, **27**, 621-628.
- Steen, H.** 1994. Low survival of long distance dispersers of the root vole, *Microtus oeconomus*. *Ann. Zool. Fennici*, **31**, 271-274.
- Sullivan, T. P. & Crump, D. R.** 1984. Influence of mustelid scent gland compounds on suppression of feeding by snowshoe hares (*Lepus americanus*). *J. Chem. Ecol.*, **10**, 1809-1821.
- Ylönen, H.** 2001. Predator odours and behavioural responses of small rodents: an evolutionary perspective. In: *Advances in Vertebrate Pest Management II*. (H.J. Pelz, P. D. Cowan and C. J. Feare.) pp. 123-138. Filander: Fürth, Germany.

## 7.CHAPTER SEVEN: DO RELEASE PROTOCOLS INFLUENCE TRANSLOCATION OUTCOMES WHEN PREDATION RISK IS LOW?

Submitted to Biological Conservation;

Moseby, K.E., Hill, B.M. and Lavery, T. (under review). Do release protocols influence translocation outcomes when predation risk is low?

Chapter 7- Moseby, K.E., Hill, B.M. and Lavery, T.H. (in prep).

Do release protocols influence translocation outcomes when predation risk is low?

Submitted to Biological Conservation June 2012

**Author Contribution**

Conceived and designed the experiments - KEM

Performed the experiments including field work-KEM, BH, T.

Analysed the data-KEM

Wrote the paper-KEM

**Author Declaration**

I approve the above author contribution statement and give permission for the paper co-authored by me to be included in the thesis by Katherine Moseby

---

Name: Brydie Hill

Signature:

Date:

Name: Tyrone Lavery

Signature:

Date: 08 June 2012

## Abstract

Translocation programs for threatened species often include elaborate release protocols designed to improve reintroduction success. The importance of reintroduction protocols may be related to factors such as predation risk and life history strategies of released animals. We compared short term reintroduction outcomes for three species with different behavioural traits into an arid environment where exotic mammalian predators were removed. Ten captive-bred and seven wild greater stick-nest rats were hard released into a 30 km<sup>2</sup> fenced exclosure with no supplementary food or water. Wild rats selected vegetation shelter sites with greater structural density than captive bred rats, travelled further from the release site and experienced lower mortality rates. Similar differences in survival were also recorded in a separate release in 1999, suggesting that although all reintroductions were successful, wild rats should preferentially be used for reintroductions. Burrowing bettongs and greater bilbies were subjected to soft and hard releases. No difference in survival was recorded but hard-release bettongs lost more weight after release and took longer to establish underground burrows leaving them exposed to increased predation risk from aerial predators. In comparison, although hard released bilbies lost weight initially, there was no difference in movement between soft and hard or wild and captive-bred bilbies and all females were carrying pouch young within 9 weeks of release. Results highlight the need for species and location-specific release protocols but suggest that using soft or hard releases or captive or wild stock may not change overall reintroduction outcomes for enclosed release sites with low predation risk. However, soft release protocols may still be warranted on ethical or containment grounds where release sites are unrestricted, in situations of high predation risk or when social, sedentary species with high shelter investment are being released.

## Key Words

Reintroduction, soft release, captive-bred, hard release, mammal, Australia

## 7.1 INTRODUCTION

Species translocations are now widely used in conservation programs throughout the world. Reintroductions are one type of translocation and are defined by the IUCN (1998) as “an attempt to establish a species in an area which was once part of its historical range, but from which it has been extirpated or become extinct”. Bajomi (2010) summarised the results of several global reintroduction reviews and found that only 11-62% of reintroduction programs are successful. On a global scale, factors influencing reintroduction success include predation, habitat condition and the size of reintroduced populations (Griffith *et al.*, 1989; Wolf *et al.*, 1996).

Most wildlife agencies now require a translocation proposal to be submitted prior to the reintroduction of native species. Translocation proposals include release protocols that are primarily designed to increase the chance of reintroduction success through minimising the

mortality of released individuals. Reintroductions can be categorised as hard or soft releases depending on whether assistance such as food, shelter, acclimatisation pens or water is provided at the time of release (Beck *et al.*, 1994). Soft releases are now a common component of many reintroduction strategies (see Soorae, 2010) and globally there is some evidence to show that the failure rate of translocations is reduced if supportive measures are undertaken (Fischer and Lindenmayer, 2000). Initial containment after release can reduce large scale movement, an activity thought to increase mortality and prevent establishment of cohesive populations (Kleimen 1989; Bright and Morris, 1994; Mateju *et al.*, 2011), although this may not apply to species that exhibit high waste accumulation attractive to predators (Banks *et al.*, 2002). Davis (1983) reported less post-release movement away from the release site in soft-released martens (*Martes americana*) compared to those that were hard released and Gedeon *et al.* (2011) found soft releasing European ground squirrels (*Spermophilus citellus*) into pre-established burrows with retention caps aided reintroduction success. Soft releases may also give animals time to adjust to their new surrounding and minimise panic movements, mortality (Bright and Morris, 1994) or stress. Teixeira *et al.* (2007) considered stress an important but rarely considered contributor to the high mortality rates reported in many translocation programmes and recommended soft releases for all animals released into the wild. In Australia, high release site fidelity may also improve survival by ensuring released animals remain in areas where exotic predators can be intensively managed. Some Australian researchers believe soft release strategies have improved mammalian reintroduction success (Southgate *et al.*, 1994, Short and Turner, 2000, Richards and Short, 2003) but have not tested this experimentally.

Conversely, other studies have found little benefit in using soft release strategies. Short (2009) reviewed the success of 73 hard and soft mammal releases in Australia and found that hard releases of mammals were typically more successful. In Western Australia, Hardman and Moro (2006) found no difference in mortality, movement or condition of hard and soft released mala (*Lagorchestes hirsutus*) and mernnine (*Lagostrophus fasciatus*) as did Moro (2001) for Lakeland downs short-tailed mice (*Leggadina lakedownensis*) released onto Serrurier Island. Griffith *et al.* (1989) and Wolf *et al.* (1998) found no significant differences in survival between soft and hard releases of mammals and birds in North America and Australia. Soft release protocols for birds have also been found to fail to reduce post-release dispersal (e.g. Clarke *et al.*, 2002) and Beck (1994) found the practice of acclimatisation was unrelated to reintroduction success of captive-born animals.

Another important part of the release strategy is the source of released animals. The IUCN reintroduction guidelines (1998) suggest that source populations should be from wild stock as long as the removal does not impact on the remaining wild population. However, evidence from previous reintroduction attempts is conflicting. Fischer and Lindenmayer (2000) found that global reintroductions using wild stock were more successful than those using captive-bred animals. Short (2009) reviewed Australian translocations and found no difference in success between mammal reintroductions using captive or wild stock as did Wolf *et al.* (1996, 1998) for North American and Australian studies.

The conflicting results found in global reintroduction reviews for the success of reintroductions using hard versus soft releases and captive versus wild stock may be due to a number of factors including different life history traits and/or the paramount influence of predation on reintroduction success. Predation is the most common cause of reintroduction failure in Australian mammal and bird programs (Fisher and Lindenmayer, 2000; Short, 2009), but not North America (Fisher and Lindenmayer, 2000). In areas where predation risk is high, soft releases and the use of wild source stock might strongly benefit reintroduction

outcomes by reducing movement and containing them within a high-intensity predator management area or giving released animals time to learn new foods or acclimatise to new surroundings without the added pressure of predation. But how important are release protocols in areas where predation risk is low? The time taken for captive-bred or hard released animals to adjust to their new environment may not differ significantly from that of wild or soft released animals or may be of little consequence in situations where predators are removed or controlled. In such cases, hard/captive-bred releases may represent a cheaper, simpler and more efficient method of release. Social, sedentary species that invest heavily in building warrens or shelter sites may benefit from soft releases more than solitary, transient species.

The influence of predation risk or behavioural traits on the success of release protocols has not been widely studied and many release protocols appear to be based on intuition, anthropomorphism or the precautionary principle (e.g. Wanless *et al.*, 2002) rather than science. Short *et al.* (1992) found that despite no supporting evidence, most macropod releases in Australia used initial containment after release. Reintroduction biology is still evolving as a science and reintroduction programs are increasingly being designed to answer research questions as well as improve the conservation status of threatened species (Griffiths and Pavajeau, 2008). Arid Recovery is a private conservation reserve in northern South Australia. Four threatened mammal species have been successfully reintroduced to a fenced enclosure where introduced feral cat (*Felis silvestris*), red fox (*Vulpes vulpes*) and native dingo (*Canis lupus dingo*) mammalian predators have been excluded (Moseby *et al.* 2011). We compared post-release behaviour, mortality, condition and movement in wild versus captive and soft versus hard released mammals in a restricted release area with low predation risk. Three mammal species were used with different behavioural traits. Results were used to recommend reintroduction protocols for the three species tested as well as more broadly for comparable release site attributes and species with similar behaviour.

## 7.2 METHODS

### 7.2.1 STUDY SITE

Established in 1997, the Arid Recovery Reserve (30°29'S, 136°53'E) is a 123 km<sup>2</sup> fenced enclosure situated 20 km north of Roxby Downs in arid South Australia (Fig. 1). A 1.8 m high wire netting fence with a curved overhang and two electric wires excludes European rabbits (*Oryctolagus cuniculus*), feral cats and red foxes (Moseby and Read, 2006). The reserve is divided into six paddocks, with introduced rabbits, cats and foxes removed from a total of 60 km<sup>2</sup>; the Main Enclosure (14km<sup>2</sup>), First Expansion (8km<sup>2</sup>), Second Expansion (8km<sup>2</sup>), and Northern Expansion (30km<sup>2</sup>).

The dominant landforms within the reserve are longitudinal orange sand dunes separated by clay interdunal swales. Dunes are generally spaced 100 m to 1 km apart. Three main habitat types are present; sandhill wattle (*Acacia ligulata*)/hopbush (*Dodonaea viscosa*) dunes, chenopod (*Atriplex vesicaria*)/(*Maireana astrotricha*) shrubland swales, and mulga (*Acacia aneura*) sandplains. Drainage is endoreic, into claypans and swamps. The Roxby Downs climate is arid, failing to reach its long term average rainfall of 166 mm in 60 % of years (Read, 1995). Rainfall is aseasonal and significant falls can occur in any month. The Roxby



Downs region has historically been used for sheep (*Ovis aries*) and cattle (*Bos taurus*) grazing.

### 7.2.2 REINTRODUCTIONS

Four locally-extinct nationally threatened mammal species, namely the greater stick-nest rat (*Leporillus conditor*), greater bilby (*Macrotis lagotis*), western barred bandicoot (*Perameles nasuta*) and burrowing bettong (*Bettongia lesueur*), have been successfully reintroduced to the Arid Recovery Reserve (Moseby *et al.*, 2011). All of these species have been extinct on the mainland of South Australia for more than 50 years (Finlayson, 1961; Copley, 1990; Kemper, 1990). Extinction is thought to have been due to a combination of predation from introduced foxes and cats, and habitat degradation from rabbits and domestic stock (Finlayson, 1962; Copley, 1990; Morton, 1990). Species were first reintroduced into the Main Exclosure between 1999 and 2001 and details of these reintroductions are outlined in Moseby *et al.* (2011).

This study outlines three experimental release protocols implemented for reintroductions of burrowing bettongs, greater bilbies and greater stick-nest rats into the Northern Expansion between October 2002 and July 2003. Firstly, wild bettongs were released in both soft and hard release treatments. Secondly, wild bilbies were hard released and results compared with hard and soft released captive bred bilbies. Lastly, captive bred and wild stick-nest rats were hard released into the Northern Expansion. All wild stock was obtained from within the Main Exclosure of the Arid Recovery Reserve. Post-release monitoring was conducted during all releases. Track counts suggested that very low densities of bilbies (<0.5 per sq km) were known to be present in the northern expansion area at the time of the experiment (Arid Recovery unpublished data). No other reintroduced species were present.

We compared post-release mortality, movement and behaviour in three species with different behavioural traits. The burrowing bettong and bilby are nocturnal animals that spend the daylight hours underground in burrows. Bilbies are solitary, transient animals and proficient diggers, regularly constructing new burrows throughout their home range (Moseby and O'Donnell, 2003). Conversely, bettongs exhibit high burrow fidelity and live in family groups down multi-entranced warren systems (Santer *et al.*, 1997). Greater stick-nest rats, whilst mainly nocturnal, are also known to be active during the day particularly in close proximity to their nests. They construct complex nests from sticks that are often woven amongst dense shrubs (Copley, 1999). Each adult female resides in a nest with her most recent young. Stick-nest rats are susceptible to aerial predators and prefer areas of thick vegetation cover for nesting and foraging.

Table 1. The table lists all animals released into the Northern Expansion, their initial characteristics, date of access to Northern Expansion and their release treatment. All animals were fitted with radiotransmitters before release.

\* = Date of release into soft release pen first, access to the rest of Northern Expansion occurred three weeks later.+ = All females had pouch young when checked 7-9 weeks after release

Species	Date	Release Type	Sex	Weight	Condition	Pouch/ Comments
Burrowing Bettong	15/10/2002	Hard	M	1668	Good	
	15/10/2002	Hard	M	1539	Fair	
	15/10/2002	Hard	M	1400	Good	
	15/10/2002	Hard	M	1442	Poor	
	15/10/2002	Hard	F	1397	Good	Inactive
	15/10/2002	Hard	F	1372	Good	Inactive
	15/10/2002	Hard	F	1460	Good	Inactive
	15/10/2002	Hard	F	1446	Good	Inactive
	15/10/2002*	Soft	F	1255	Good	Inactive
	15/10/2002*	Soft	F	1292	Good	Inactive
	15/10/2002*	Soft	F	1528	Good	Inactive
	15/10/2002*	Soft	M	1382	Fair	
	15/10/2002*	Soft	M	1643	Good	
	15/10/2002*	Soft	M	1635	Good	
Greater Bilby	29/4/2003	Hard Wild	F	1159		Inactive+
	29/4/2003	Hard Wild	F	952		Inactive+
	29/4/2003	Hard Wild	M	1194		
	29/4/2003	Hard Wild	M	1088		
	8/4/2003	Hard Captive	F	958		Inactive+
	8/4/2003	Hard Captive	F	1084		Inactive+
	8/4/2003	Hard Captive	M	960		
	8/4/2003	Hard Captive	M	958		
	8/4/2003*	Soft Captive	F	1040		Inactive+
	8/4/2003*	Soft Captive	F	990		Inactive+
	8/4/2003*	Soft Captive	M	1265		
	8/4/2003*	Soft Captive	M	1030		
Stick-nest Rat	2/7/2003	Captive	M	290	Excellent	
	2/7/2003	Captive	M	362	Excellent	
	2/7/2003	Captive	F	283	Excellent	
	2/7/2003	Captive	F	363	Excellent	
	2/7/2003	Captive	M	359	Excellent	
	2/7/2003	Captive	F	314	Excellent	
	2/7/2003	Captive	F	309	Excellent	
	2/7/2003	Captive	F	360	Excellent	
	2/7/2003	Captive	M	326	Excellent	
	2/7/2003	Captive	M	385	Excellent	
	5/7/2003	Wild	M	330	Good	
	5/7/2003	Wild	F	120	Fair	Juvenile
	5/7/2003	Wild	F	130	Fair	Juvenile
	14/7/2003	Wild	F	165	Good	
	16/7/2003	Wild	F	244	Good	
	24/7/2003	Wild	F	275	Poor	
	27/7/2003	Wild	M	115	Fair	Juvenile

### *7.2.3 SOFT VERSUS HARD RELEASE BURROWING BETTONGS*

On the 15 October 2002, 14 bettongs were transferred to the Northern Expansion of the reserve from the Main Enclosure (Table 1). Animals were captured in cage traps baited with peanut butter and rolled oats and were weighed, sexed and checked for body and reproductive condition. Bettongs were randomly allocated to either soft or hard release treatments with equal numbers of each sex in each treatment. Eight (4M,4F) of the animals were hard released directly into the Northern Expansion without any supplementary shelter, food or water. Hard release animals were released next to, but outside of, the soft release pen and were radiotracked daily for the first week and then every second day for three weeks after release. The remaining six animals (3M, 3F) were placed in a 2 ha soft release pen for three weeks. Prior to release, one metre deep burrows were dug throughout the pen using a long handled shovel and dense piles of branches were constructed providing immediate shelter for the animals. Soft release animals were provided with supplementary food and water for three weeks after release after which time holes were cut in the pen and animals allowed access to the Northern Expansion. Soft release animals were radiotracked daily for the first week after their release into the release pen and then every second to third day for the following two weeks. After holes were cut in the netting they were radiotracked daily for a week after the release pen was open and then every second day for three weeks. Soft release animals gained access to the Northern Expansion three weeks after the hard release animals.

During daily radiotracking, the location of each animal was marked using a GPS and its position described as either in a burrow or on the surface under vegetation. Linear distances between locations on consecutive days were calculated using a GPS. Hard release bettongs were recaptured at two and four weeks post release using baited cage traps. Soft release bettongs were recaptured two weeks after being placed in the soft release pen and then two and four weeks after the release pen was open. All animals were weighed and checked for body and reproductive condition. The change in body weight over time was compared between soft and hard released bettongs using a two way repeated measures ANOVA. Soft and hard released bettong weights were also compared between capture and at two and four weeks after access to the Northern Expansion. For soft release bettongs this was actually at five and seven weeks post release due to the three weeks spent in the release pen.

The daily movement of bettongs was calculated as the distance between consecutive diurnal locations. Once the bettong was recorded in the same diurnal location for two consecutive sampling days it was assumed that the animal had established itself in a burrow. The time until burrow establishment in the Northern Expansion and the distance of the established burrow from the release pen were compared between soft and hard released animals. For soft release animals, the time taken until burrow fidelity was recorded once they had left the soft release pen. The distance moved on the first night of release was also compared using the first night after the release pen was opened for soft release animals. Data were first tested for normality using the Shapiro Wilk test which is more appropriate for small sample sizes than the Kolmogorov Smirnov test. Q-Q (quantile) plots were also visually assessed for normality. Data that were normally distributed were analysed using independent samples Student t-tests. Levene's test for equality of variances was used and if significant then equal variances were not assumed. Data that were not normally distributed were transformed using  $\ln+1$  or compared using nonparametric Mann Whitney U tests.

#### 7.2.4 SOFT VERSUS HARD RELEASE OF CAPTIVE AND WILD GREATER BILBIES

On 8 April 2003, eight captive-bred bilbies from Monarto Zoological Park, 70 km East of Adelaide, South Australia, were released into the Northern Expansion under two release protocols; a hard release of four animals (2M,2F) directly into the Northern Expansion and a soft release of four animals (2M,2F), into the 2ha release pen with supplementary food, water and pre-existing burrows (Table 1). Three weeks after release, the soft release animals were allowed access to the Northern Expansion through holes cut in the sides of the release pen. Simultaneously on 29 April 2003, four wild-born bilbies (2M,2F) were translocated from the Main Enclosure to the Northern Expansion using the hard release technique. Captive and wild hard released animals were radiotracked for up to a total of 34 days. All bilbies were radiotracked daily for the first 11 days after access to the Northern Expansion and the daily distances moved between diurnal fixes and the distance travelled from the release pen were compared between the soft captive, hard captive and hard wild treatments. The home range size of each animal was estimated using the 100 percentage minimum convex polygon method and compared between treatments using a one way ANOVA and transformation of data using  $\ln+1$ . Soft release bilbies were recaptured at one and three weeks after release into the soft release pen and then again at seven to nine weeks post release after they had had access to the Northern Expansion for four to six weeks. Hard release animals were recaptured at one and seven to nine weeks post release. Animals were captured in cage traps and their weight and reproductive condition assessed before being released back into their burrow. The distance moved between fixes was compared between treatments as well as the maximum distance moved from the release pen over the first 11 days. Statistical analyses were as outlined previously for the burrowing bettong.

#### 7.2.5 CAPTIVE BRED VERSUS WILD STICK-NEST RATS

The initial release of stick-nest rats into the Main Enclosure of the Arid Recovery Reserve in 1999 is summarised in Moseby *et al.* (2011). However, additional information on the mortality of 6 captive-bred rats and 44 wild rats released into the Main Enclosure in April 1999 is presented here. Captive-bred rats were sourced from the greater stick-nest rat breeding program at Monarto Zoological Park. Rats were captured in their pens in the morning of release and transported to the site by car (approximately 7 hr drive). Wild rats were obtained from Reevesby Island, South Australia and captured using Elliott traps and hand held nets. Rats were captured in the evening or early morning and transported using the same method as captive rats but with a slightly longer travel time of approximately 9 hrs. Both captive and wild rats were released simultaneously into the same release area in the early evening and radiotracked daily for one month after release.

The reintroduction of rats into the Northern Expansion of the Arid Recovery Reserve occurred in July 2003. Twenty captive-bred greater stick-nest rats from Monarto Zoological Park were transported by car (1 hr drive) to the Adelaide Zoo for veterinary examinations on the 1 July 2003. Four greater stick-nest rats were rejected for release due to the presence of cataracts or watermarks on the eyes, ten of the remaining 16 rats (5M,5F) were fitted with radio-collars (Titley Scientific, Australia). The 16 rats were transported to the Arid Recovery Reserve on 2 July 2003 in wooden transport boxes lined with straw and released into the Northern Expansion of the reserve at 20:00 h.

Wild-reared rats were trapped using small and large Elliott traps and treadle wire cage traps set in the Main Enclosure throughout July 2003. Traps were set at 20 nest sites and in areas of high rat activity. Lactating females or previously tagged animals were re-released at the capture site and only new animals were used in the study. Seven rats, five females and two males, were radiocollared and relocated to the Northern Expansion between 5 and 27 July 2003. Due to the difficulty with capturing wild-reared greater stick-nest rats, three of the relocated rats were subadults.

All animals, captive-bred and wild, were weighed and tagged before being released into the Northern Expansion on a sand dune primarily vegetated with sandhill wattle shrubs, *Acacia ligulata*, hummocks of dune canegrass, *Zygochloa paradoxa*, and ruby saltbush, *Enchylaena tomentosa*. After release, rats were located daily by radiotelemetry and a GPS was used to record the position of each rat. Additional information about the site including substrate, vegetation species and the density of cover provided by the vegetation were also recorded. Cover density was assessed by placing a 1.5 m pole painted with 5cm black and white bands horizontally through the shrub at a height of approximately 30cm. The number of black bands that could not be observed at a height of 30cm within the shrub was recorded as a percentage of the total number of bands known to be within the shrub. For example, if no light penetrated the shrub at ground level and the observer could not see any of the bands on the pole then it received a cover density of 100%.

If an animal was found dead the carcass and surrounding area was carefully inspected for signs of predators. Animals were recaptured three weeks after release and re-weighed and their condition assessed. At three weeks post release, the distance from each rat's location to the release site was compared between captive and wild rats using an independent samples Student's t- test. The distance moved between successive diurnal fixes was also compared. Data were analysed using similar methods to those for burrowing bettongs.

After three weeks, the vegetation composition and cover were calculated within an area defined by the 90% minimum convex polygon method of the combined diurnal fixes of all rats. Two perpendicular 550 m line transects were placed through the 90% minimum convex polygon area and the shrub species and their cover density recorded at 10 m intervals. Ephemeral vegetation and grasses were not included. Results were used to assess the proportion of different shrub species and cover categories available as shelter and feeding sites. The actual proportion of diurnal fixes recorded at shrub species and cover densities was compared to the proportion of expected fixes at shrub species and cover densities if selection was due to chance. Data were compared using Chi-squared tests and contingency tables. Due to low sample sizes, data from individual rats were pooled within captive and wild treatments.

## 7.3 RESULTS

### 7.3.1 HARD VERSUS SOFT RELEASED BURROWING BETTONGS

No bettongs died during the five to seven weeks of post release monitoring in the Northern Expansion. All hard release animals located on the morning after release were found on the surface sheltering in thick vegetation such as canegrass hummocks or *Acacia ligulata* shrubs. All hard release bettongs were recorded down burrows within 7 days of release. All burrows except one were situated on dunes, most were excavated old disused rabbit warrens but some had been dug in soft sand. In comparison, only 25% of soft release bettongs were found on the surface the day after they were released into the release pen. Of the four soft release animals that were found down burrows, three used the pre-excavated burrows provided and one used an old bilby burrow. After a few days, bettongs moved into burrows that they dug themselves. Once the release pen was opened and bettongs allowed access to the Northern Expansion, all of the soft release bettongs were found in burrows on the day after they left the release pen. Soft and hard released bettongs were not found sharing burrows suggesting that soft release bettongs did not take advantage of burrows previously excavated by burrowing bettongs.

A repeated measures ANOVA conducted on weight data collected up until 5 weeks post release found no significant difference between the weights of bettongs over time ( $F_{1,839}=1.250$ ,  $P=0.303$ ) or between treatments (soft vs hard  $F_1=1.348$ ,  $P=0.268$ ). There was a significant time by treatment interaction ( $F_{1,839}=13.308$ ,  $P<0.001$ ) suggesting that the change in weight over time was different for soft and hard released animals. Soft release animals increased weight in the release pen before losing weight once the release pen was opened (Fig.1). In comparison, hard released animals lost weight initially before gaining weight. Interestingly, average weights of hard release bettongs were still lower than their release weight at 5 weeks post release (Fig. 1). Conversely, weights of soft release animals had not dropped below their release weight 5 weeks after the release pen was opened. No pouch young were recorded at the time of release or at subsequent checks at 2, 5 or 7 weeks post release but spoor counts and subsequent trapping revealed that the population later expanded and remains extant in 2012.

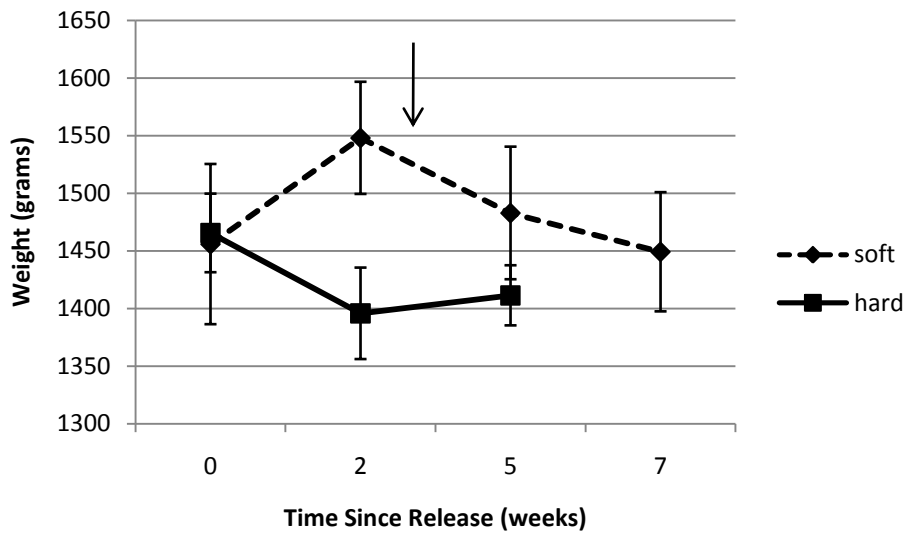


Fig. 1: Weights of hard (n=8) and soft (n=6) released burrowing bettongs. The arrow indicates when release pen was first opened and soft release animals allowed access to the northern expansion. Bars denote one standard error.

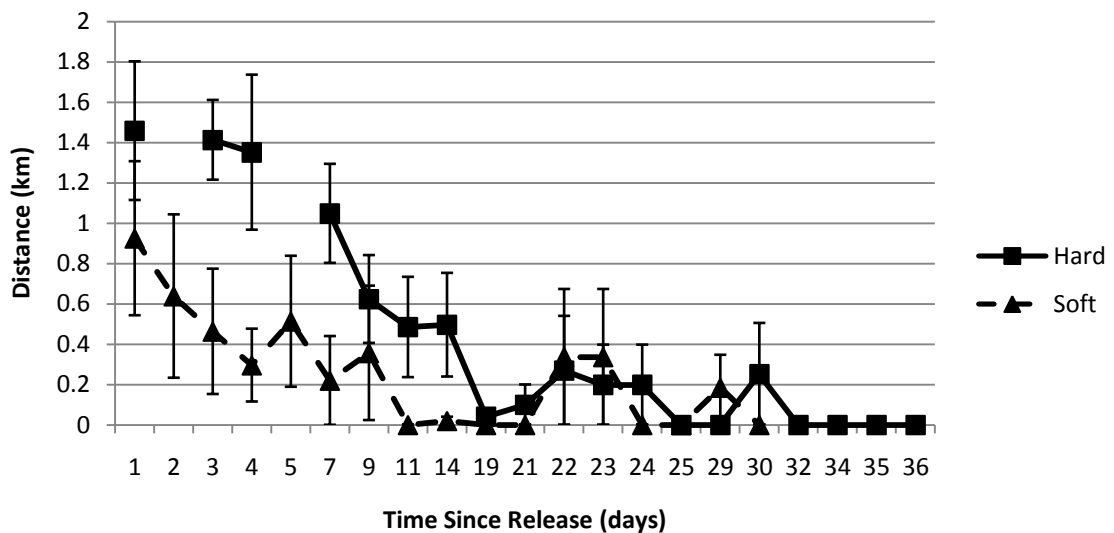


Fig. 2: The distance between successive daily fixes for soft and hard released bettongs after access to the Northern Expansion. Soft release bettongs were kept in a release pen for two weeks prior to release and were thus released 2 weeks after hard release animals. Bars denote one standard error.

The average distance moved by hard release bettongs on the night of release was 1.46 km (SE=0.34) compared with 0.925 km (SE=0.38) for soft release animals on the night after the release pen was opened (Fig. 2). However, large variation between individuals rendered this difference not significant ( $t_{12}=-1.187$ ,  $p=0.258$ ). Movement between successive diurnal fixes declined to zero at around 11 days after the release pen was opened for soft release animals and approximately 19 days after release for hard released bettongs (Fig. 2).

The average time until successive daily movements ceased and animals exhibited burrow fidelity in the Northern Expansion was 13.1 days for hard released bettongs and 3.6 days for soft release animals. There was a significant difference between soft and hard release animals in the number of days taken until burrow fidelity was achieved ( $t_{10,2}=3.457$ ,  $p=0.006$ ). Two of the three soft release females did not leave the release pen for more than a week after it was opened and their first move resulted in burrow fidelity. One female temporarily moved back into her old burrow inside the release pen for one night after a week living outside the release pen.

After burrow fidelity was achieved, there was a trend towards soft release animals settling closer to the release pen (average 1.14 km SE=0.41) than hard release animals (1.83 km SE=0.09) but this was not significant ( $t_{5,47}=1.61$ ,  $p=0.16$ ).

### *7.3.2 HARD VERSUS SOFT RELEASED BILBIES*

No bilbies were known to have died after release into the Northern Expansion. Low sample sizes and difficulties recapturing bilbies rendered statistical comparisons of weight difficult. All released bilbies were a similar weight at release but soft release animals increased weight after release whilst hard released animals lost weight initially (Fig.3). Both wild and captive hard release animals had returned to their release weight seven to nine weeks after release. By comparison, soft release animals increased in weight and maintained the increase after the release pen was opened. All female bilbies regardless of soft or hard release had pouch young when captured at 7 to 9 weeks after release. Spoor counts indicate that the population increased and remains extant in 2012.

Bilbies were difficult to locate after release and radiotracking data were haphazard despite attempting to locate each bilby every day. On average, bilbies moved up to 1.5 km each day between diurnal burrow fixes (Fig.4). The number of fixes collected from each bilby over the first 11 days after access to the Northern Expansion varied from 3 to 9 (average 5.4, E=0.53).



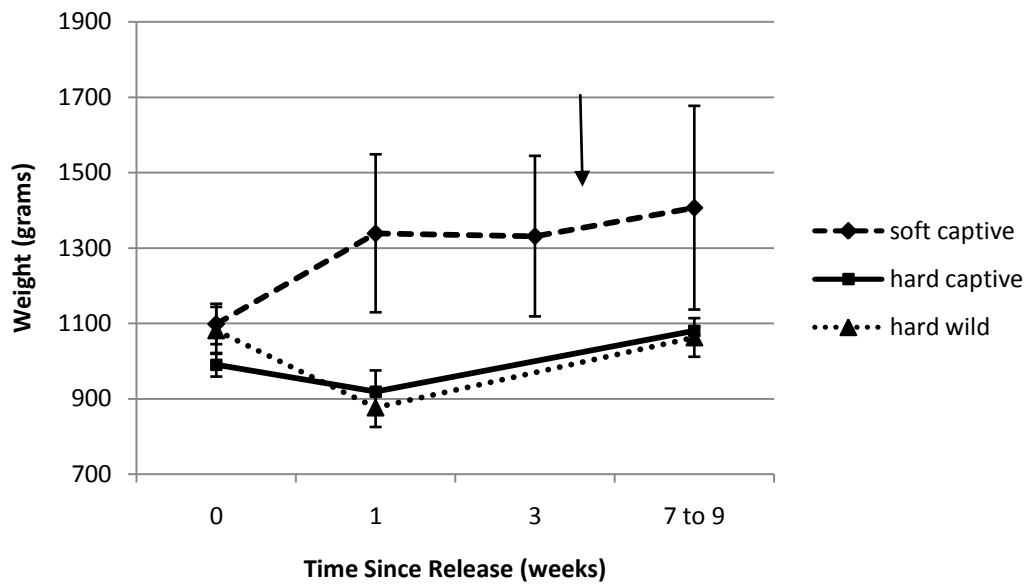


Fig. 3: Average weights of soft, hard wild and hard captive bilbies released into the Northern Expansion in 2003. Although all 4 bilbies in each treatment were captured and weighed at release and 1 week after release, only three soft, three hard captive and four hard wild bilbies could be recaptured at seven to nine weeks for reweighing. Bars denote 1 standard error. Arrow indicates when release pen was opened.

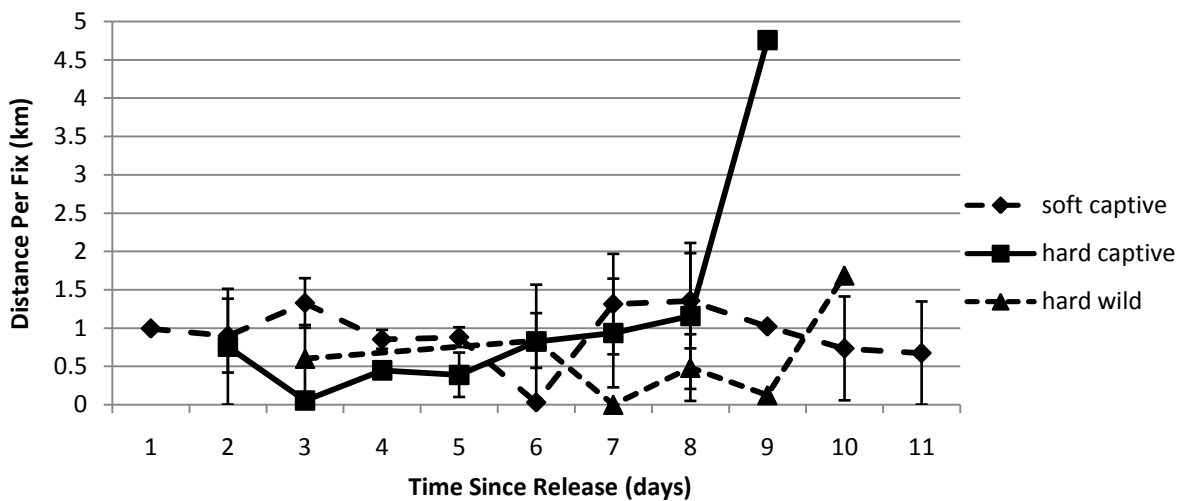


Fig.4: Average distances moved between radiotracking fixes for soft and hard released bilbies after release into the Northern Expansion. For soft release animals, time since release refers to when the soft release pen was opened and animals allowed access to the Northern Expansion. Bars denote 1 Standard Error, points without bars are single individuals.

To account for the differences in sampling effort, the total distance moved by each bilby over the first 11 days was divided by the number of fixes to calculate the distance moved per fix. The average distance moved per fixes was 0.53 km (SE=0.22) for hard wild bilbies, 0.93 km (SE=0.72) for hard captive bilbies and 0.94 (SE=0.24) for soft captive bilbies. Distances were compared between treatments using a one way ANOVA and ln+1 transformation and there was no significant difference between the distance moved for soft, hard captive or hard wild

bilbies ( $F_2=0.371$ ,  $p=0.7$ ). When all hard release bilbies were pooled and compared with soft release animals no significant difference was detected ( $F_1=0.105$ ,  $P=0.429$ ).

The maximum distance moved from the release pen in the first 11 days after release was 4.5 km by a hard released captive-bred male and the smallest distance was 0.29 km by a hard released wild caught male. There was no significant difference in maximum distance moved between hard captive, hard wild and soft release bilbies ( $F_2=0.337$ ,  $p=0.722$ ) or between soft and pooled hard release bilbies ( $F_1=0.743$ ,  $p=0.409$ ).

There was no significant difference in the home range size of wild, hard captive or soft captive bred bilbies ( $F_2=0.045$ ,  $P=0.965$ ). Home range varied considerably from 2.8 ha (female) to 614 ha (male). Time to reach burrow fidelity was not compared as bilbies are known to move burrows regularly.

### *7.3.3 CAPTIVE-BRED VERSUS WILD STICK-NEST RATS*

During the initial release of stick-nest rats in 1999, two of the six (33.3%) captive bred rats released into the Main Exclosure died at 6 and 15 days after release compared with two of the 19 (10.5%) radiocollared wild caught rats that died at 7 and 8 days post release. Intact rat carcasses were found dead on the ground in the open or under bushes with no sign of predation. The cause of death was thought to be stress or malnutrition.

Four of the ten radiocollared captive bred rats released into the Northern Expansion died within 3 weeks of release compared to no deaths recorded in the seven wild rats. Three of the captive bred rats died within four days of release, two from predation by birds of prey and the third from unknown causes. The fourth rat was observed on the surface lethargic and panting 12 days after release after which time radiotracking indicated that it remained down a large bettong warren. The carcass was dug up from inside the warren a week later.

Recapturing stick-nest rats after release was problematic due to collar failure and whilst all of the captive bred rats were recaptured, only two of the seven wild-bred rats were able to be recaptured. The six captive bred rats that survived after release weighed an average of 325.8 g ( $SE=12.5$ ) at release, dropping to 277.5 g ( $SE=11.9$ ) at three weeks post release and recovering slightly to 292 g ( $SE=11.6$ ) at five weeks post release. Comparatively, wild rats were lighter, the average weight of the four adult wild rats was 253.5 g ( $SE=34.4$ ) at release. Only two rats from the wild reared sample were recaptured, one was a juvenile that had increased from 120 g to 235 g in four weeks. The other adult wild rat had lost 10 g over the same time period. Due to the low sample size of wild-reared rats as a result of radio-collar failure it was not possible to statistically compare weight change data.

Movements of the surviving six captive bred stick-nest rats and the seven wild rats were compared during the first three weeks after release. All captive bred rats were radiotracked each day ( $n=21$ ) but wild rats moved greater distances and could only be located on average 17 times ( $SE=1.16$ ) over the three week period. To standardise the data, the total cumulative distance moved between daily fixes over the three week period was divided by the number of fixes to obtain a standard distance moved per fix. Wild rats moved far from the release site initially and then remained at the same location whereas the captive bred rats continued to move around relatively close to the release site until tracking ceased. There was no significant difference in the distance moved per fix between wild and captive rats ( $t_{11}=1.175$ ,  $p=0.265$ ).

However, at three weeks post release, wild rat shelter sites were significantly further from the release site than captive rats ( $t_{11}=2.664$ ,  $p=0.022$ ). All six surviving captive stick-nest rats were situated in shelter sites within 125 m of the release point (average 65.0, SE=17.3) whilst six of the seven wild rats were located more than 120 m from the release site (average = 728.7 m, SE=365.6). Two of the wild rats settled 1,549 and 2,581 m from the release site respectively and the three juvenile wild rats recorded the shortest distance moved from the release site.

There were 223 diurnal locations recorded for captive rats and 121 for wild rats over the first month after release. Burrowing bettong warrens were used on 18 occasions by captive rats and 10 occasions by wild rats and were excluded from the analysis of plant species and cover selection. The selection of plant species for shelter sites was not random for either captive ( $\chi^2=41.719$ , d.f=4,  $P<0.001$ ,  $n=205$ ) or wild ( $\chi^2=55.43$ , d.f=4,  $P<0.001$ ,  $n=111$ ) rats. There was also a significant difference in the plant species selected for shelter sites between captive bred and wild rats ( $\chi^2=25.970$ , d.f=3,  $n=316$ ,  $P<0.001$ ). Captive bred rats selected a greater proportion of *Zygochloa paradoxa* (sandhill canegrass), *Acacia ligulata* and *Enchylaena tomentosa* (ruby saltbush) than expected according to chance and wild rats selected a greater proportion of *Z. paradoxa* than expected.

Similarly, both captive ( $\chi^2=169.269$ , d.f=4,  $n=205$ ,  $P<0.001$ ) and wild ( $\chi^2=107.37$ , d.f=4,  $n=111$ ,  $P<0.001$ ) rats selected denser than average shelter sites. There was also a difference between the density of shelter sites selected by wild and captive rats ( $\chi^2=56.208$ , d.f=4,  $n=316$ ,  $P<0.001$ ) with wild rats choosing shelter sites with the thickest cover (80-100%) (Fig. 5) and captive rats using a wider range of cover densities including sparser shrubs (Fig. 6).

Nest and spoor counts conducted annually up until 2012 indicate that populations of stick-nest rat remain established in both the Main Enclosure and Northern Expansion.

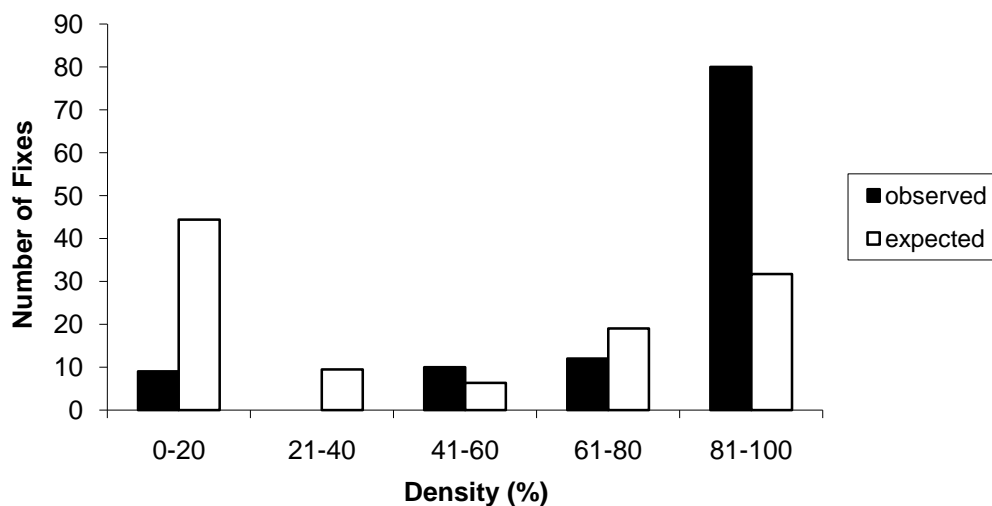


Fig. 5: The number of diurnal fixes of wild rats recorded at each of the shelter density categories compared with the expected fixes if shelter choice was due to chance.

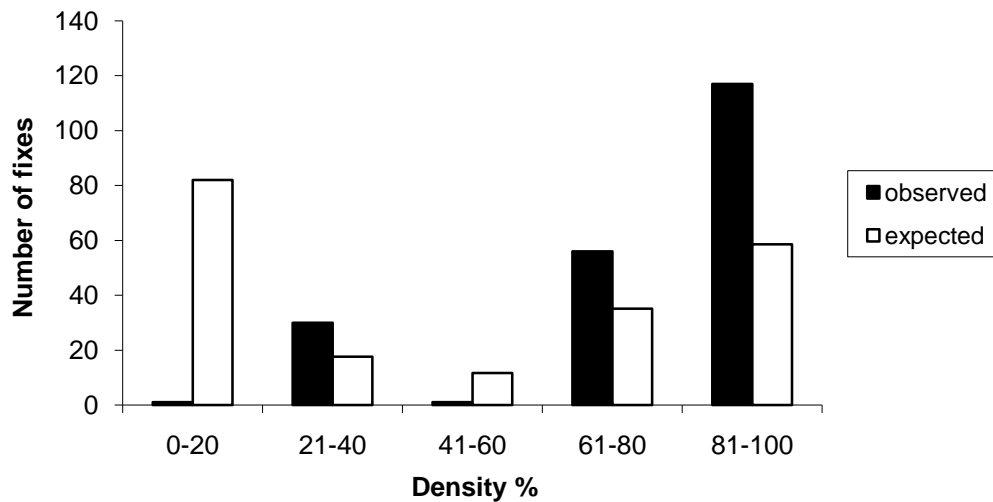


Fig. 6: The number of diurnal fixes of captive bred rats recorded at each of the shelter density categories compared with the expected fixes if shelter choice was due to chance.

## 7.4 DISCUSSION

Differences in mortality after release were only observed in captive-bred versus wild stick-nest rats with captive rats exhibiting higher mortality than wild Arid Recovery or wild island rats after release. Wild-bred animals have been reported as having higher survival rates than captive-bred individuals in other Australian studies (bridled nailtail wallabies, *Onychogalea fraenata*, Pople *et al.*, 2001) and during international reviews (Griffith *et al.*, 1989; Beck *et al.*, 1994). Snyder *et al.* (1996) reported that severe behavioural deficiencies in some captive-bred animals were responsible for high reintroduction failure particularly in species that rely on learned behaviour. Behavioural differences were observed in our study with shorter movement from the release site and poorer selection of shelter sites recorded in captive-bred rats. Our results are consistent with Bright and Morris (1994) who also recorded shorter movements in captive bred dormice compared with wild born animals. Poor shelter selection may have contributed to the higher predation rates on captive-bred rats and may also affect survival rates during periods of extreme temperatures as stick-nest rats are known to suffer summer die offs in arid areas (Bolton and Moseby, 2004). In comparison, no differences in mortality or behaviour of captive-bred and wild bilbies were recorded, consistent with results from Short *et al.* (1992) who found no clear difference in reintroduction success of macropods in Australia using wild or captive-bred stock.

Inconsistencies between post release survival and behaviour of captive bred and wild rats and bilbies could be explained by predation risk and patterns of learning. Snyder *et al.* (1996) hypothesised that behavioural problems would be less common in captive-bred species that had low levels of parental care and that captive-bred releases would be most effective in areas with no predators. Our results support these findings, as bilbies have lower levels of parental care than stick-nest rats. The predation risk for rats was higher than bilbies as rats are more susceptible to aerial predators, but the low predation risk by exotic cats and foxes at our release site ensured that releases of both captive-bred species were successful. Other captive-

bred Australian species have also been successfully reintroduced to predator-free areas (e.g. Morris, 2000; Moro, 2003). The predation risk and level of parental care should be assessed for each species before deciding if the use of captive-bred stock is desirable. In situations of low predation risk, the use of captive-bred stock may not significantly affect post release survival or reintroduction success.

There was no difference in mortality between soft and hard released bettongs or bilbies, similar to Short and Turner (2000) who found no difference in burrowing bettong survival when comparing animals released into familiar and unfamiliar environments within a predator free refuge. Globally, Griffith *et al.* (1989) and Wolf *et al.* (1996) also found no significant differences in survival between soft and hard releases of mammals and birds. Our results and reviews of reintroduction programs suggest that the benefits of soft releases may be overstated and that soft releases may only be required in certain circumstances as outlined below.

#### 7.4.1 HIGH PREDATION RISK

As in our study, many other reintroduction studies have found no difference between soft and hard released animals where these were conducted in situations of low predation risk (Short *et al.*, 1992; Campbell and Croft, 2001; Hardman and Moro, 2006). Similarly, successful hard release reintroductions are often into islands or fenced reserves, also areas with low predation risk (Short and Turner, 2000; Moro, 2003; Moseby and O'Donnell, 2003; Moseby *et al.*, 2011; Richards and Short, 2003). Short (2009) found hard released animals in Australian reintroductions had greater survival than soft release animals. This conclusion is counter intuitive but may be explained by the high number of hard releases conducted into areas with little or no predation risk such as islands and fenced sanctuaries. Soft releases may not be necessary when predation risk is low but may be required when predation risk is high. Hard released animals, however, may be more unfamiliar with their environment and find it difficult to source appropriate food and shelter, exposing them to higher predation risk.

#### 7.4.2 ETHICAL GROUNDS

Soft releases may also be justified on ethical grounds. Textiera *et al.* (2007) highlighted the importance of stress in shaping reintroduction outcomes and suggested that all reintroductions should be soft releases. Sub-clinical stress and stress accumulation are both thought to impact on an individual's ability to find food, socialise and avoid predation (Textiera *et al.*, 2007). Stress may be difficult to measure and may have latent impacts on reintroduction outcomes (Rosatte *et al.*, 2002). In our study there was some evidence that hard released bettongs suffered more stress than soft released animals. Hard released bettongs took longer to establish permanent burrows, were more likely to be on the surface during the day, and unlike soft released bettongs, their weights dropped below release weight. Practitioners tend to focus on the overall success of reintroductions at a population scale rather than the wellbeing of the individuals involved. Even though the use of a release pen may not improve population survival in areas where predation risk is low, pens may minimise stress and improve conditions for the individual resulting in a more ethical release.

### 7.4.3 UNBOUNDED RELEASE SITES

Soft releases may be useful for initial containment at unrestricted release sites. There was a trend towards longer movements in hard released animals in our study and the differences in movement are likely to have been under represented as the Northern Expansion is only 30 km<sup>2</sup> and the maximum distance that could be moved from the release pen was 4 km. Without the presence of the fence hard released bilbies and bettongs likely would have travelled much greater distances. Hard releases of bettongs outside the Arid Recovery Reserve in 2008 resulted in movements of up to 18 km from the release point (Moseby *et al.*, 2011) and several male burrowing bettongs released into Heirisson Prong in W.A. were recorded moving more than 10 km and up to 21 km from the release site (Short and Turner, 2000). Males that disperse large distances from the release site are unlikely to contribute to the population and there is evidence that dispersing individuals have higher mortality rates than non-dispersers (Kleimen, 1989; Bright and Morris, 1994; Steen, 1994; Norrdahl and Korpimaki, 1998). Some studies on other species have also found large scale movements of hard released animals (Davis, 1983; Christensen and Burrows, 1994) and a soft release may retain animals close to the release site where predator control can be intensified. In contained release sites such as islands or fenced mainland reserves, soft releases may not be necessary. Hard released dighters all stayed within 400 m of the release site on Escape Island (Moro, 2003) and Hardman and Moro (2006) found no difference in site fidelity between hard and soft releases of mala and mernnine onto a fenced peninsula. The use of a release pen for bettong reintroductions is likely to be advantageous in areas where the release site boundaries are unrestricted and intensive predator management is conducted around the release area.

### 7.4.4 SOCIAL OR SEDENTARY LIFE HISTORY STRATEGIES

Unlike bettongs, release protocols or source populations appeared to have little influence on the survival, health, movement or reproductive output of bilbies. Inter-specific differences may be due to a number of different natural history traits. Bilbies are a relatively transient species that move burrows regularly suggesting that bilbies continually encounter new habitats. This trait may mean they find translocations less stressful than sedentary species such as bettongs that exhibit higher site fidelity. Bilbies are also expert diggers and can quickly establish a new burrow after release. In comparison bettongs invest significant energy into building permanent burrows rendering them more likely to attempt to move large distances after release searching for their home burrow exposing them to increased predation risk. In species that do not invest heavily in building permanent burrows or nests, soft releases may not be necessary. Thompson *et al.* (2001) found no difference in movements between soft and hard released mountain sheep (*Ovis Canadensis*) a species that does not build shelters. Social species such as bettong may also find translocation more stressful than bilbies which are a solitary species. There are many examples where soft releases have enhanced reintroduction success in social species. In France, female rabbits survived better when acclimatised using a soft release pen (Letty *et al.*, 2000) with improvements attributed to sex-specific social behaviour. Wimberger *et al.* (2010) reported that a reintroduction of the gregarious rock hyrax in South Africa failed due to high dispersal and suggested containment of family groups in a soft release pen prior to release would improve reintroduction outcomes. Shier (2006) used soft releases of family groups of black tailed prairie dogs to improve

reintroduction success and found higher survival than non-family groups. Kleiman (1989) suggested that in social species, dispersal after release may be partly because translocated animals often lack familiarity with individuals at the release site. Reintroduction success of the social European ground squirrel (*Spermophilus citellus*) was increased when soft release pens were used to prevent panic dispersal, establish a new social order and adjust to new food resources (Mateju *et al.*, 2011). Hardman and Moro (2006) found no difference in movement or condition between hard and soft released mala which is a solitary species that does not invest significant energy in building shelters. The use of release pens in social, sedentary species that spend significant time establishing permanent shelters may assist reintroduction success by keeping family groups intact, allowing time for communal burrow or den establishment and reducing stress.

#### 7.4.5 SUPPLEMENTATION

Finally, soft releases may be useful when supplementing reintroduced populations for genetic reasons. Our study, where animals were released into an area with no or low density of conspecifics, may be very different to releasing animals into occupied habitat. Genetic supplementation has failed at some release sites due to the high density of inhabitants preventing new stock from establishing. Genetic supplementation of brush-tailed bettongs, *Bettongia penicillata*, on Wedge Island and subsequent releases of bilbies at Arid Recovery failed after all new founders died after release (pers. obs). Previously released animals may have established defendable territories or have depleted food resources and may exclude newcomers. Other advantages of soft release pens are that they can facilitate immediate post-release monitoring and also allow natural dispersal if the pen mesh size is large enough to allow juveniles to escape. Richards and Short (2003) found juvenile western barred bandicoots could disperse from the refuge through the wire mesh.

Providing supplementary food and water after release may assist with preventing initial post-release weight loss and is a common method used by many reintroduction practitioners (e.g. Southgate *et al.*, 1994). However, the benefit of short term weight gain versus the cost of short term weight loss was not clear in our study; both soft and hard released bilbies were carrying pouch young at seven to nine weeks post release and by then hard released animals had regained their lost weight. The weight loss of hard released bettongs was not significant and the weight gained by soft release bettongs was lost within a few weeks of food being removed. In the absence of predation pressure, supplementary feeding appeared to have no influence on reintroduction success. Hardman and Moro (2006) also found no significant difference between weights of hard and soft released Australian mala and mernnine. Wanless *et al.* (2002) believed a soft release assisted reintroduction success of the Aldabra rail (*Dryolimnas aldabranus*) by allowing birds to regain weight lost during transit and increase energy reserves. However, these conclusions were based on the precautionary principle rather than experimentation and the researchers themselves highlighted the need for developing criteria to determine when soft releases are necessary. Similar weight losses in wild and captive-bred bilbies after release suggests that wild animals were no better at detecting new food sources or that stress affected them equally and that both groups would respond to supplementary feeding.

Logistical constraints meant that some treatment groups did not have access to the Northern Expansion area simultaneously. Ideally soft release bettongs would have been placed in the

pen three weeks prior to the hard release so that both release groups were allowed access to the Northern Expansion at the same time. However, if this had occurred then cutting holes in the release pen may have allowed hard release animals to access the pen on the night of release. Additionally, soft release bettongs did not use any burrows previously dug by hard release animals so it is unlikely that these minor time differences influenced project results.

## 7.5 CONCLUSIONS

In our study, post release survival, movement, weight loss and behaviour differed between release protocols for some species highlighting the importance of developing species-specific release strategies. The key indicator of a successful reintroduction is survival and all releases were ultimately successful suggesting that release strategies may have limited influence on reintroduction outcomes when predation risk is low. The release site in our study closely resembled an island release scenario where exotic predators are often removed and the release area is contained. Extrapolation of results suggests that release strategies are also likely to have little impact on the outcome of island releases in the absence of predators and may help explain why island releases generally have a much higher success rate than mainland ones (Short *et al.* 1992). However, other indicators of reintroduction success are important both from an ethical and productivity perspective. Soft releases may be less stressful (Teixeira *et al.* 2007), increase site fidelity and lead to faster doubling times. However, soft-release pens can also have negative influences due to overcrowding and intraspecific aggression in males (Lyne 1982, Letty *et al.* 2000, Richards and Short 2003). Studies in the wild (Neumann 1999) suggest that space reduction can increase aggression and disrupt normal social behaviour particularly in solitary species (Lyne 1982). The potential negative impact on animal behaviour coupled with higher labour and funding costs suggests that soft releases should be targeted to situations where they are likely to be beneficial such as where release site boundaries are unrestricted, where predation risk is high or in social, sedentary species that invest heavily in their shelter sites. Researchers undertaking reintroduction reviews are advised to include predation risk and life history strategy as covariates when analysing the factors influencing reintroduction success.

## 7.6 ACKNOWLEDGEMENTS

This study was supported by Arid Recovery, a conservation initiative assisted by BHP Billiton, The University of Adelaide and the South Australian Department for Environment and Natural Resources. Many volunteers assisted with field work including Ruth Coates, Bree Galbraith, John Read, Michael Foate, Zoos S.A. and Earthwatch volunteers. Funding was provided by Arid Recovery, Earthwatch, Friends of Arid Recovery and the Nature Foundation S.A. Monarto Zoological Park kindly bred captive animals for release. Reintroductions were conducted under ethics approval from the Wildlife Ethics Committee, South Australia and permits from the South Australian Department for Environment and Natural Resources.



## 7.7 REFERENCES

- Bajomi, B., 2010. Reintroduction of endangered animal species: complementing the IUCN guidelines. Review protocol. (Accessed the 16th February 2011). Centre for Environmental Science, Eötvös Loránd University of Sciences, Budapest, Hungary, Pp. 10.  
<http://www.environmentalevidence.oeg/Documents/Draftprotocols86.pdf>.
- Banks, P.B., Norrdahl, K., Korpimäki, E., 2002. Mobility decisions and the predation risks of reintroduction. *Biol. Cons.* 103, 133–138.
- Beck, B. B., Rapaport, L. G., Price, M. R. S., Wilson, A. C., 1994. Reintroduction of captive-born animals. in: Olney, P.J.S., Mace, G. M., Feistner, A.T.C. (Eds.), *Creative Conservation: Interactive Management of Wild and Captive Animals*. Chapman & Hall, London, U.K., pp. 265-286
- Bright, P. W., Morris, P. A., 1994. Animal translocations for conservation: performances of dominance in relation to release methods, origin and season. *J. Applied Ecol.* 31, 699-708.
- Bolton, J., Moseby, K.E., 2004. The activity of Sand Goannas *Varanus gouldii* and their interaction with reintroduced Greater Stick-nest Rats *Leporillus conditor*. *Pac. Cons. Biol.* 10, 193-201.
- Bright, P.W., Morris, P.A., 1994. Animal translocation for conservation: performance of doormice in relation to release methods, origin and season. *J. Applied Ecol.* 31, 699–708.
- Campbell L., Croft, D.B., 2001. Comparison of hard and soft release of hand reared eastern grey kangaroos in: Martin, A., Vogelnest, L. (Eds), *Veterinary Conservation Biology Wildlife Health and Management in Australasia*, Proceedings of International Joint Conference, Taronga Zoo, Sydney, Australia, pp. 173–180.
- Christensen, P., Burrows, N., 1994. Project desert dreaming: experimental reintroduction of mammals to the Gibson Desert, Western Australia. In: Serena, M. (Ed.) *Reintroduction Biology of Australian and New Zealand Fauna*. Surrey Beatty & Sons, Sydney, Australia, pp. 199-207.
- Clarke, R.H., Boulton, R.L., Clarke, M.F., 2002. Translocation of the socially complex Black-eared Miner *Manorina melanotis*: a trial using hard and soft release techniques. *Pac. Cons. Biol.* 8, 223–234.
- Copley, P., 1999. Natural histories of Australia's stick-nest rats, genus *Leporillus* (Rodentia:Muridae). *Wildl. Res.* 26, 513-539.
- Davis, M. H., 1983. Post-release movements of introduced marten. *J. Wildl. Mgmt.* 47, 59-66.
- Finlayson, H.H., 1961. On central Australian mammals, Part IV. The distribution and status of central Australian species. *Rec. S.A. Mus.* 41, 141-191.
- Fischer, J., Lindenmayer, D.B., 2000. An assessment of the published results of animal relocations. *Biol. Cons.* 96, 1-11.

- Gedeon, C.I., Vászi, O., Koósz, B., Altbäcker V., 2011. Morning release into artificial burrows with retention caps facilitates success of European ground squirrel (*Spermophilus citellus*) translocations. *Eur J Wildl Res.* doi:10.1007/s10344-011-0504-3
- Griffith, B., Scott, J.M., Carpenter, J.W., Reed, C., 1989. Translocation as a species conservation tool: status and strategy. *Science* 245, 477-480.
- Griffiths, R.A., Pavajeau, L., 2008. Captive breeding, reintroduction and the conservation of amphibians. *Cons. Biol.* 22, 852-861.
- Hardman, B., Moro, D., 2006. Optimising reintroduction success by delayed dispersal: Is the release protocol important for hare-wallabies? *Biol. Cons.* 128, 403-411.
- IUCN, 1998. Guidelines for reintroductions. Prepared by the IUCN / SSC Re-introduction Specialist Group., IUCN, Gland, Switzerland and Cambridge, UK.
- Kemper, C., 1990. Status of bandicoots in South Australia. in (Eds. J. H. Seebeck, J.H., Brown, P.R., Wallis, R.I., Kemper, C.M. (Eds.), *Bandicoots and Bilbies*. Surrey Beatty & Sons, Sydney, Australia, pp. 67-72.
- Kleiman, D.G., 1989. Reintroduction of captive mammals for conservation. Guidelines for introducing endangered species into the wild. *Bioscience* 39, 152-161.
- Letty, J., Marchandea, S., Clobert, J., Aubineau, J., 2000. Improving translocation success; an experimental study of anti-stress treatment and release method for wild rabbits. *Anim. Cons.* 3, 211-219.
- Lyne, A.G. and CSIRO Division of Wildlife and Rangelands Research (1982). The bandicoots *Isoodon macrourus* and *Perameles nasuta*: their maintenance and breeding in captivity. in: Evans, D.D. (Ed.), 'The Management of Australian Mammals in Captivity'. Zoological Board of Victoria, Melbourne, Australia, pp. 47-52.
- Matějů, J., Říčanová, S., Poláková, S., Ambros, M., Kala, B., Matějů, K., Kratochvíl, L., 2011. Method of releasing and number of animals are determinants for the success of European ground squirrel (*Spermophilus citellus*) reintroductions. *Eur. J. Wildl. Res.* 58, 473-482.
- Moro, D., 2001. Update of translocation of Thevenard Island mice to Serrurier Island, Western Australia: success and excess. *Reintro.News* 20, 17-18.
- Moro, D., 2003. Translocation of captive-bred dibblers, *Parantechinus apicalis* (Marsupialia: Dasyuridae) to Escape Island, Western Australia. *Biol. Cons.* 111,305-315.
- Morris, K., 2000. Fauna translocations in Western Australia 1971-1999: an overview. in: Gillen, J.S., Hamilton, R., Low, W.A., Creagh, C. (Eds.), *Biodiversity and the Reintroduction of Native Fauna at Uluru-Kata Tjuta National Park*. The Bureau of Rural Sciences, Kingston ACT, pp. 64-74.
- Morton, S.R., 1990. The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proc. Ecol. Soc. Aust.* 16, 201-213.

- Moseby, K.E., O'Donnell, E., 2003. Reintroduction of the greater bilby, *Macrotis lagotis* (Reid) (Marsupialia:Thylacomyidae), to northern South Australia: survival, ecology and notes on reintroduction protocol. *Wildl. Res.* 30, 15-27.
- Moseby, K.E., Read, J.L., 2006. The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biol. Cons.* 127, 429-437.
- Moseby, K.E., Read, J.L., Paton, D.C., Copley, P., Hill, B.M., Crisp, H.M., 2011. Predation determines the outcome of 10 reintroduction attempts in arid Australia. *Biol. Cons.* 144, 2863-2872.
- Neumann, D. R., 1999. Agonistic behavior in harbor seals (*Phoca vitulina*) in relation to the availability of haul-out space. *Mar. Mam. Sci.*, 15, 507-525.
- Norrdahl, K., Korpimäki, E., 1998. Does mobility or sex of voles affect risk of predation by mammalian predators? *Ecology* 79, 226-232.
- Pople, A.R., Lowry, J., Lunding-Jenkins, G.W., Clancy, T.F., McCallum, H.I., Sigg, D., Hoolihan, D., Hamilton, S., 2001. Demography of bridled nailtail wallabies translocated to the edge of their former range from captive and wild stock. *Biol. Cons.* 102, 285-299
- Read, J.L., 1995. Recruitment characteristics of the white cypress pine (*Callitris glaucophylla*) in arid South Australia. *Rangel. J.* 17, 228-240.
- Richards, J.D., Short, J., 2003. Reintroduction and establishment of the western barred bandicoot *Perameles bougainville* (Marsupialia: Peramelidae) at Shark Bay, Western Australia. *Biol. Cons.* 109, 181-195.
- Rosatte, R., Hamr, J., Ranta, B., Young, J. & Cool, N. 2002. Elk restoration in Ontario, Canada: infectious disease management strategy, 1998-2001. *Annals of the New York Academy of Sciences, Domestic Animal/Wildlife Interface: Issue for Disease Control, Conservation, Sustainable Food Production, and Emerging Diseases*, 969, 358e365.
- Santer, U, Short, J., Turner, B., 1997. Social organisation and warren use of the burrowing bettong, *Bettongia lesueur* (Macropodoidea: Potoroidae). *Wildl. Res.* 24, 143-157.
- Short, J., Bradshaw, S.D., Giles, J., Prince, R.I.T., Wilson, G.R., 1992. Reintroduction of macropods (Marsupialia: Macropodoidea) in Australia, a review. *Biol. Cons.* 62, 189-204.
- Short, J., Turner, B., 2000. Reintroduction of the burrowing bettong *Bettongia lesueur* (Marsupialia: Potoroidae) to mainland Australia. *Biol. Cons.* 96, 185-196.
- Short, J., 2009. The characteristics and success of vertebrate translocations within Australia. Australian Government Department of Agriculture, Fisheries and Forestry, Canberra.
- Snyder, N.F.R., Derrickson, S.R., Beissinger, S.R., Wiley, J.W., Smith, T.B., Toone, W.D., Miller, B.J., 1996. Limitations of captive breeding in endangered species recovery. *Cons. Biol.* 10, 338-348.
- Soorae, P.S. ed., 2010. Global re-introduction perspectives: Additional case-studies from around the globe. IUCN/ SSC Re-introduction Specialist Group, Abu Dhabi, United Arab Emirates.

- Southgate, R., Bellchambers, K., Romanow, K., Whitfield, S., 1994. Reintroduction of the Greater Bilby A Field Guide. Final Report to World Wide Fund for Nature (Australia), Northern Territory, Australia.
- Steen, H. 1994. Low survival of long distance dispersers of the root vole, *Microtus oeconomus*. *Ann. Zool. Fennici* 31, 271-274.
- Teixeira, C. P., C. S. de Azevedo, M. Mendl, C. F. Cipreste, Young, R.J., 2007. Revisiting translocation and reintroduction programmes: the importance of considering stress. *Anim. Behav.* 73, 1-13.
- Thompson, J.R., Bleich, V.C., Torres, S.G., Mulcahy, G.P., 2001. Translocation techniques for mountain sheep: does the method matter? *The S.West. Nat.* 46, 87-93.
- Wanless, R.M., Cunningham, J., Hockey, P.A.R., Wanless, J., White, R.W., Wiseman, R., 2002. The success of a soft-release reintroduction of the flightless Aldabra rail (*Dryolimnas [cuvieri] aldabranus*) on Aldabra Atoll, Seychelles. *Biol. Cons.* 107, 203-210.
- Wimberger, K., Downs, C.T., Perrin, M.R., 2009. Two unsuccessful re-introductions of rock hyraxes (*Procavia capensis*) into a reserve in the KwaZulu-Natal Province, South Africa. *Sth Afric. J. Wildl. Res.* 39, 192-201.
- Wolf, C.M., Garland, T Jr., Griffith, B., 1998. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts *Biol. Cons.* 86, 243-255.
- Wolf, C.M., Griffith, B., Reed, C., Stanley, A., 1996. Avian and Mammalian Translocations: Update and Reanalysis of 1987. *Cons. Biol.* 10, 1142-1154.

## 8. CHAPTER EIGHT: KEEP ON COUNTING: THE IMPORTANCE OF LONG TERM POST RELEASE MONITORING IN REINTRODUCTION PROGRAMS

To be submitted to Austral Ecology;

Keep on counting: the importance of long term post-release monitoring in reintroduction programs.

## Abstract

Post-release monitoring of translocated species is an integral part of reintroduction programs but intensive long term monitoring is rarely conducted due to financial or logistical reasons. We investigated the factors influencing post-release population fluctuations in four species of reintroduced mammal in arid South Australia. Population indices derived from track activity of IUCN- listed greater bilbies, burrowing bettongs, greater stick-nest rats and western barred bandicoots were measured along sand dunes up to 39 times over 12 years after release into the Arid Recovery Reserve. The effect of environmental variables on these population indices was compared. Interestingly, even over a 12 years timeframe, the abundance of three species was most strongly influenced by time since release, suggesting that reintroduced populations may take decades to stabilise. Greater bilby activity declined when rainfall within the previous nine months dropped below average whilst changes in stick-nest rat track abundances were influenced by high summer temperatures. Western barred bandicoot activity was positively correlated with winter rainfall in the 6 months prior to sampling. Results from long term monitoring were used to suggest improvements in future reintroduction strategies such as releasing stick-nest rats in autumn, providing larger or multiple release areas for bilbies to prevent population extinction during droughts, implementing overpopulation strategies for burrowing bettongs within five years of release and releasing western barred bandicoots immediately after significant post-summer rainfall events. We suggest three phases of post release monitoring. Phase one involves initial intensive monitoring of individuals to determine short term reintroduction success. Phase two requires seasonal post-release population monitoring conducted until reintroduced populations are no longer influenced by time since release in order to accurately assess reintroduction outcomes. This second phase of monitoring should continue until populations have completed several cycles of environmentally-induced fluctuations in order to accurately ascertain the influence of environmental variables on the fecundity, mortality and population trajectories of reintroduced species. The time frame required to reach phase two monitoring milestones may extend into several decades, particularly when release sites are large, founder populations small, and in desert environments where climatic patterns are unpredictable. Once populations have demonstrated their propensity to equilibrate to prevailing resource levels, the third phase of monitoring could involve a reduction in the frequency of monitoring to levels expected in natural wild populations.

## 8.1 INTRODUCTION

Animal translocations are an increasingly important tool in conservation programs in Australia (Short 2009) and overseas (Seddon *et al.* 2007). Translocations include introductions, reintroductions and supplementations (Sutherland *et al.* 2010) and aim to establish or re-establish a species in an area where it is either absent or declining. Mammals, reptiles, birds, plants and amphibians have all been successfully translocated but the success rate varies considerably according to variables such as species, habitat, predation and release size (Griffith *et al.* 1989, Fischer and Lindenmayer 2000, Short 2009). In Australia, the majority of failed mammal reintroduction attempts are attributed to predation by introduced red foxes and feral cats (Fischer and Lindenmayer 2000, Short 2009, Moseby *et al.* 2011).

Determining the fate of reintroduced populations is arguably the most important component of a translocation program and the key to improving global reintroduction outcomes (Armstrong and Seddon 2008; Sutherland *et al.* 2010). The IUCN reintroduction guidelines (IUCN 1998) state that post-release monitoring is an essential part of the reintroduction process and many researchers have emphasised its importance in reintroduction programs (Fischer and Lindenmayer 2000, Seddon *et al.* 2007). Post-release monitoring of translocated populations is often conducted in two phases. The initial intensive monitoring period usually lasts several weeks or months and involves radiotracking or observing a few individuals daily for a short period after release (for examples see Soorae 2010, Moseby *et al.* 2011). Monitoring frequency and intensity then usually declines or ceases altogether. The second medium to long term phase is usually characterised by intermittent or opportunistic monitoring using a population abundance estimate. Sutherland *et al.* (2000) suggest the minimum acceptable standard for post-release monitoring of bird translocations is monitoring at 1 and 5 years after release and at 10 and 15 years for very long lived species. However, they also indicate that post-release monitoring is the greatest challenge in translocation programs. Fischer and Lindenmayer (2000) found few global reintroduction programs followed a strict post-release monitoring program, making it difficult to determine the outcome of translocation projects. Only some programs have the resources to implement intensive post-release monitoring with the result that sporadic genetic resampling or the occasional opportunistic survey of translocated populations is conducted to document their continued survival or extinction.

In addition to determining the fate of reintroduced species, we suggest that medium to long term monitoring is required to improve our understanding of reintroduction biology. Intensive post-release monitoring can provide new insights into the ecology or behaviour of rare species, with information used to improve future translocation outcomes. This is particularly important in reintroduction programs where the species is locally extinct and little information exists on its former habits. The need for long term monitoring, and more ecological information on re-introduced species, were common themes espoused in global reintroduction case studies compiled by the IUCN (Soorae 2010).

To investigate the benefits of intensive long term post-release monitoring, we recorded the abundance of four reintroduced mammal populations (greater bilby (*Macrotis lagotis*), greater stick-nest rat (*Leporillus conditor*), western barred bandicoot (*Perameles bougainville*), and burrowing bettong (*Bettongia lesueur*)) in arid South Australia up to 39 times after release over a 12 year period. Abundance was measured using a passive activity index along transects within the Arid Recovery Reserve; a fenced exclosure where introduced cats, foxes and rabbits have been removed (Read *et al.* 2011). Fluctuations in abundance were compared with a range of intrinsic (time since release) and extrinsic (rainfall, season, Indian Ocean Dipole) factors to determine the ecological drivers influencing activity in each species. Results were used to determine the importance of long term intensive post-release monitoring and demonstrate how monitoring can inform and improve future release strategies.

## 8.2 METHODS

### 8.2.1 STUDY SITE

Established in 1997, the Arid Recovery Reserve (30°29'S, 136°53'E) is a 123 km<sup>2</sup> fenced enclosure situated 20 km north of Roxby Downs in arid South Australia (Fig. 1). A 1.8 m high wire netting fence with a curved overhang excludes rabbits, cats and foxes (Moseby and Read 2006). The Reserve is divided into six paddocks with introduced rabbits, cats and foxes removed from four paddocks (Main Enclosure, First Expansion, Northern Expansion, Second Expansion) totalling 60 km<sup>2</sup>.

The dominant landforms within the reserve are longitudinal orange sand dunes separated by clay interdunal swales. Dunes are generally spaced 100 m to 1 km apart. Three main habitat types are present; sandhill wattle (*Acacia ligulata*)/hopbush (*Dodonaea viscosa*) dunes, chenopod (*Atriplex vesicaria*)/(*Maireana astrotricha*) shrubland swales, and mulga (*Acacia aneura*) sandplains. Drainage is endoreic, into claypans and swamps. The Roxby Downs climate is arid, failing to reach its long term average rainfall of 166 mm in 60 % of years (Read 1995). Rainfall is aseasonal and significant falls can occur in any month. The Roxby Downs region, has historically been used for sheep (*Ovis aries*) and cattle (*Bos taurus*) grazing.

### 8.2.2 REINTRODUCTIONS

Four locally extinct threatened mammal species, namely the greater stick-nest rat, greater bilby, western barred bandicoot and burrowing bettong, have been successfully reintroduced to the reserve (Moseby *et al.* 2011). The former three species became extinct on the mainland in South Australia in the 1930's (Kemper 1990) with the burrowing bettong surviving until the 1960's in the far north of the state. Extinction is thought to have been due to a combination of predation from introduced foxes and cats and habitat degradation from rabbits and domestic stock (Morton 1990). Species were first reintroduced into the Main Enclosure, a 14 km<sup>2</sup> paddock where rabbits, cats and foxes were eradicated in 1999 (Read *et al.*, 2011). Main Enclosure reintroductions occurred between 1999 and 2001, either into a small 10 ha release pen or directly into the Main Enclosure. After a few months, dispersal holes were cut in the sides of the release pen and animals allowed free access to the rest of the Main Enclosure. Details of Main Enclosure reintroductions are outlined in Moseby *et al.* (2011). Once established in the Main Enclosure, animals were gradually transferred to the First and Northern Expansion paddocks between August 2001 and November 2005 (Table 1). Western barred bandicoots dispersed naturally from the Main Enclosure to the First Expansion paddock without assisted translocations and some greater stick-nest rats also likely dispersed through the fences. The Second Expansion was kept free of reintroduced species to act as a control area. During 2005, the gates between the Main, First and Northern Expansions were opened to allow free movement between the enclosures.



### 8.2.3 POPULATION MONITORING

All reintroduced species were initially monitored using radiotelemetry for up to 12 months after release. To investigate long term trends in abundance, activity levels were measured using track (animal spoor) counts (Table 2, Fig. 1). Track counts involved walking along longitudinal sand dunes early in the morning and recording animal tracks that crossed a 1 m wide path. To ensure consistency, an animal's track was counted each time it entered and then left the path. Between 2000 and 2012, the number of tracks of greater stick-nest rat, greater bilby, burrowing bettong and western barred bandicoot were recorded along two transects within each of the Main, First and Northern Paddocks, totalling 6 transects. Transects were situated more than 2 km apart to ensure independence. Transects ranged from 2.4 to 10 km in length (Table 2) and were gradually established in each paddock as species were reintroduced. Transects were conducted 2-4 times a year, totalling up to 39 sampling periods between 2000 and 2012. Transects within a paddock were conducted concurrently in more than 98% of cases but different paddocks were often sampled on different days. Between 2000 and 2005, the track transects were monitored on the morning after a windy day and a still night, to ensure only fresh tracks from a single night were counted. After this time the population of reintroduced species increased to the degree that it became necessary to clear old tracks the day prior to sampling by dragging a 1 m steel bar and chain behind a quadbike.

Table 1: The number of individuals and dates of release for species reintroduced into the three exclosures in the Arid Recovery Reserve. Subtotals refer to the numbers released at each separate release date. Access was either through direct release or gradual immigration (I). For species that were first placed in release pens within the exclosures, dates and numbers refer to the date and number of animals in the pen when it was opened and animals first allowed access to the exclosure.

Species	Main Exclosure		First Expansion		Northern Expansion	
	Date	Total (subtotals)	Date	Total (subtotals)	Date	Total (subtotals)
greater sick-nest rat	Apr/Jun 1999	91 (51/40)	Aug2001/Feb2002	17 (15/2)	Jul2003	19
burrowing bettong	Feb/Sept 2000	30 (10/20)	Aug2001	8	Oct2002	27
greater bilby	May 2000	9	Nov2001-Apr2002	15	July/Sept 2002/Apr 2003	18 (3/3/12)
western barred bandicoot	Apr 2002	18	Nov2008	I	Nov 2005	6

The number of tracks recorded per transect for each species was converted to tracks per kilometre and compared with a number of intrinsic and extrinsic variables. Animal spoor counts are a passive activity index and are likely to be correlated with changes in activity as well as abundance. More accurate methods of population estimate such as capture-mark-recapture and distance sampling were not implemented due to significant interspecific differences in trapping effectiveness and the difficult in directly observing cryptic species.

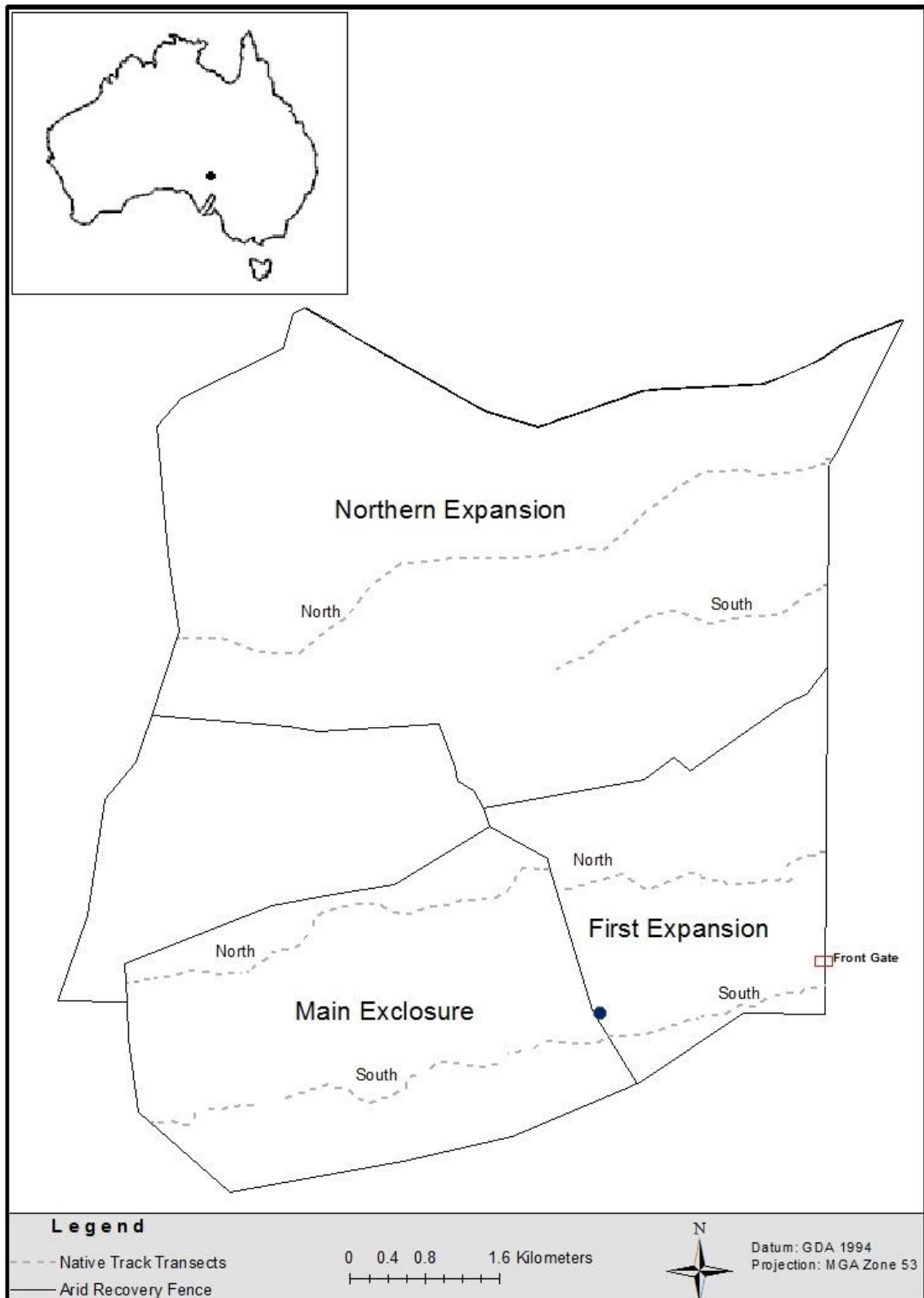


Fig. 1: The location of track transects used to monitor population abundance during the study.

Table 2: The track transects used to monitor the activity of reintroduced species after release from 2000 to 2012. The two transects within each enclosure were pooled for statistical analyses.

Transect	Length	First sampled	Last sampled	No. sampling periods
Main Enclosure North	5.2	11/5/2000	8/2/2012	39
Main Enclosure South	5.4	11/5/2000	8/2/2012	39
First Expansion North	3.1	6/4/2002	8/2/2012	30
First Expansion South	2.4	6/4/2002	8/2/2012	30
North Expansion North	10	13/3/2003	8/2/2012	32
North Expansion South	4	13/3/2003	8/2/2012	32

#### 8.2.4 VARIABLES

The following variables were compared with the tracks of each species recorded per km to investigate factors influencing the abundance of species after release.

##### **8.2.4.1 Time Since Release**

The number of months since release into each enclosure was calculated from the date of the first release (Table 1). Time since release varied between enclosures and between species. The western barred bandicoot dispersed naturally from the Main Enclosure into the First Expansion during 2008. Bandicoot tracks were first recorded in December 2008 but not in the preceding sampling period in August 2008. The time of release was therefore arbitrarily set as November 2008.

##### **8.2.4.2 Month**

The month of sampling was used to compare changes in abundance in different calendar months.

##### **8.2.4.3 Season**

Sampling sessions were pooled within seasons, based on the month of sampling. Summer sampling sessions were those conducted in December, January and February, autumn sessions were in March, April and May, winter sessions were June, July and August and spring sampling sessions were conducted in September, October and November.

##### **8.2.4.4 Moon Phase**

The percentage of the full moon was recorded for the evening of the sampling date. Additionally, for each sampling date the percentage fullness of the moon was categorised into the following moon phases; full (80-100% full), waxing (20-80% full and moon waxing), waning (20-80% full and moon waning) and new moon (0-20% full). Cloud cover was not considered.

#### **8.2.4.5 Temperature**

The maximum daily temperature of the day preceding sampling was obtained from the National Weather Bureau ([www.bom.gov.au](http://www.bom.gov.au)). The weather station was located at Olympic Dam, located 10 km from the Arid Recovery Reserve. Maximum temperature on the day preceding sampling was used as all four reintroduced species are nocturnal and most likely to be influenced by the maximum temperature recorded on the day prior to the night of activity.

#### **8.2.4.6 Cumulative monthly rainfall residual**

The cumulative monthly residual rainfall was recorded starting from January 1997, two years before the first species was reintroduced. The long term monthly average rainfall is 13.8 mm and actual monthly totals were used to determine the long term rainfall deficit or surplus.

#### **8.2.4.7 Rainfall**

Track activity was compared with rainfall recorded at the Arid Recovery Reserve over different time periods prior to each sampling event. The rain gauge was located in the First Expansion, between 0.5 and 5 km from all monitoring transects. Rainfall periods included 1-3, 4-6, 7-9, 10-12, 13-15, 16-18 and 19-21 months prior to sampling. Cumulative total rainfall recorded in the intervals 3, 6, 9 and 12 months prior to sampling was also included.

#### **8.2.4.8 Summer versus winter rainfall**

The rainfall recorded in the cooler months (Apr-Sept) and the warmer months (Oct-Mar) was analysed separately for each of the rainfall time periods and cumulative rainfall intervals outlined above. For each time period or cumulative rainfall interval prior to sampling, the total amount of rainfall recorded was separated into amounts that fell in the cooler or warmer months.

#### **8.2.4.9 Indian Ocean Dipole**

The Indian Ocean Dipole (IOD) is a coupled ocean and atmosphere phenomenon in the equatorial Indian Ocean that affects the climate (Saji *et al.* 1999) and some biological responses (Read *et al.* 2012) of arid Australia. The weekly Indian Ocean Dipole Index (NINO2 SST index) was averaged over the 12 months prior to each sampling session and compared with track activity. IOD Index data was obtained from the website [www.bom.gov.au](http://www.bom.gov.au).

### *8.2.5 DATA ANALYSIS*

#### **8.2.5.1 Multivariate analysis**

Time series representing the number of tracks per kilometre were used as indicators of abundance. Density estimates were pooled across transects within exclosures for analysis. We used generalised additive models (gam; Wood 2011) with a gaussian variance function to model temporal trends and the additive effects of environmental covariates on the density of each species. The track density variable was log-transformed for analysis to satisfy model assumptions (the greater bilby and stick-nest rat densities included zeros, so the minimum density value for each species was added to all density values prior to transformation). We used low order spline terms (i.e. constrained maximum degrees of freedom to 3 per term) to allow the models to capture any nonlinear relationships between the continuous environmental variables and track density.

Due to the time series nature of the data, we calculated the Durbin-Watson statistic (Durbin and Watson 1971) to check for evidence of temporal autocorrelation in the residuals of gam models. Where there was temporal autocorrelation, we fitted generalized additive mixed models (gamm; Wood 2004) with a continuous first-order autoregressive term to explicitly account for the correlation. Exploratory analysis was conducted with a larger set of exploratory predictors and used to reject parameters that were not important in order to minimise complexity and reduce the number of models.

We developed a candidate set of models for the relationships between track density and time since release and environmental covariates. All models retained the variable time since release to account for temporal trends in density and the interaction between time and enclosure. Due to the available sample sizes, we allowed a maximum of three additional variables in the most complex models in the candidate set. These included rainfall, maximum temperature and the Indian Ocean Dipole climate indicator. Rainfall variables were strongly correlated with each other, so we substituted rainfall variables (representing cumulative totals over the 3, 6 and 9 months preceding sampling and the 4-6 and 7-9 months preceding sampling) into the most complex model one at a time (longer lags were examined in exploratory analysis but were discarded prior to analysis). Similarly, the residual rainfall index was strongly correlated with time since release because of the consistent drought years over much of the period since reintroductions occurred. We used Akaike's information criterion corrected for finite sample size, AICc (Sugiura 1978), to rank the models in the candidate set, and inferences were based on the highest-ranked AICc model. All statistical analyses were performed using R 2-15.0 (R Development Core Team 2012). Plots represent the partial effects of environmental covariates and show 95% confidence intervals.

The summary output from the highest-ranked model was used to determine the coefficients for the slopes of increase in track abundance over time since release. These rates of increase over time (month) were presented for each species in each enclosure to enable interspecific growth comparisons.

### **8.2.5.2 Univariate Analysis**

Some relationships between variables were investigated further using linear regression. To investigate the influence of summer temperatures on the decline in track abundance of stick-nest rats, the proportional change in track abundance over the summer months (December to February inclusive) was compared with a number of temperature parameters. The proportional change was calculated using the change in track density between the last sampling session before summer and the sampling session closest to the end of summer. The proportional change was compared to the maximum summer temperature, number of days with a maximum temperature of 40°C or hotter, the hottest maximum temperature recorded and the number of consecutive days with a maximum temperature of 40°C or higher. Only track abundance within the Main Enclosure was included in the analysis as stick-nest rats have been present in this enclosure longest.

A linear regression was also used to investigate the relationship between founder size and the rate of increase since release. The association between IOD (averaged over 12 months preceding sampling) and cumulative rainfall recorded in the 12 months prior to sampling was also compared using linear regression.

## 8.3 RESULTS

### 8.3.1 RAINFALL

Rainfall during the study period was erratic with below average rainfall recorded in 6 of the years of post-release monitoring (Fig. 2). Two drought periods were identified in the 12 years between 1998 and 2010 based on severe rainfall deficiencies, a 12 month deficiency from January 2002 to January 2003 when only 43 mm of rain were recorded for the period (average of 3.3 mm per month) and an 18 month deficiency from May 2007 to October 2008 when 74 mm of rain was recorded (average of 4 mm per month) (Moseby *et al.* 2011).

### 8.3.2 TRACK ABUNDANCE

Track abundance of all four species increased within the three exclosures after release and all species remained extant in 2012 (Figs.3-6). Similar trends were observed within the three separate exclosures with burrowing bettongs increasing steadily and reaching the highest track abundance (up to 275 tracks per km). Stick-nest rats generally had the lowest track abundance of the four species (<40 tracks per km, Figs.3-6). Greater bilbies and western barred bandicoots both exhibited similar fluctuations after release occasionally reaching track abundances of up to 100 tracks per kilometre. The sharp temporary decline in track abundance observed in February 2009 occurred when the hottest maximum temperature on the day preceding sampling was recorded (46.5 °C).

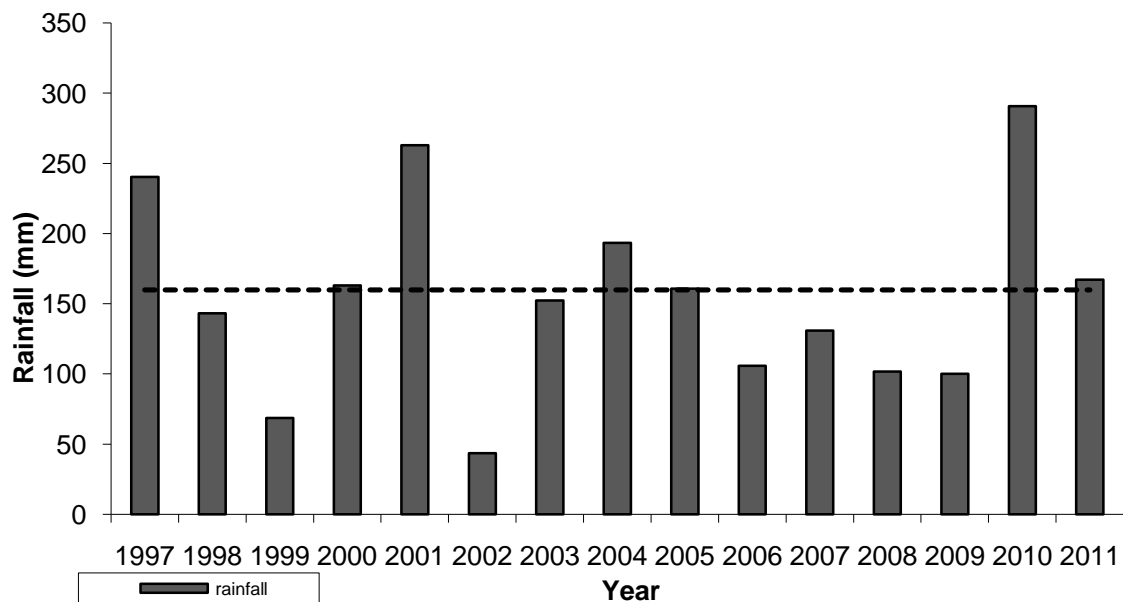


Fig. 2: Annual rainfall recorded before and during the study period. The dotted line indicates the annual average.

### Main Exclosure (14 km<sup>2</sup>)

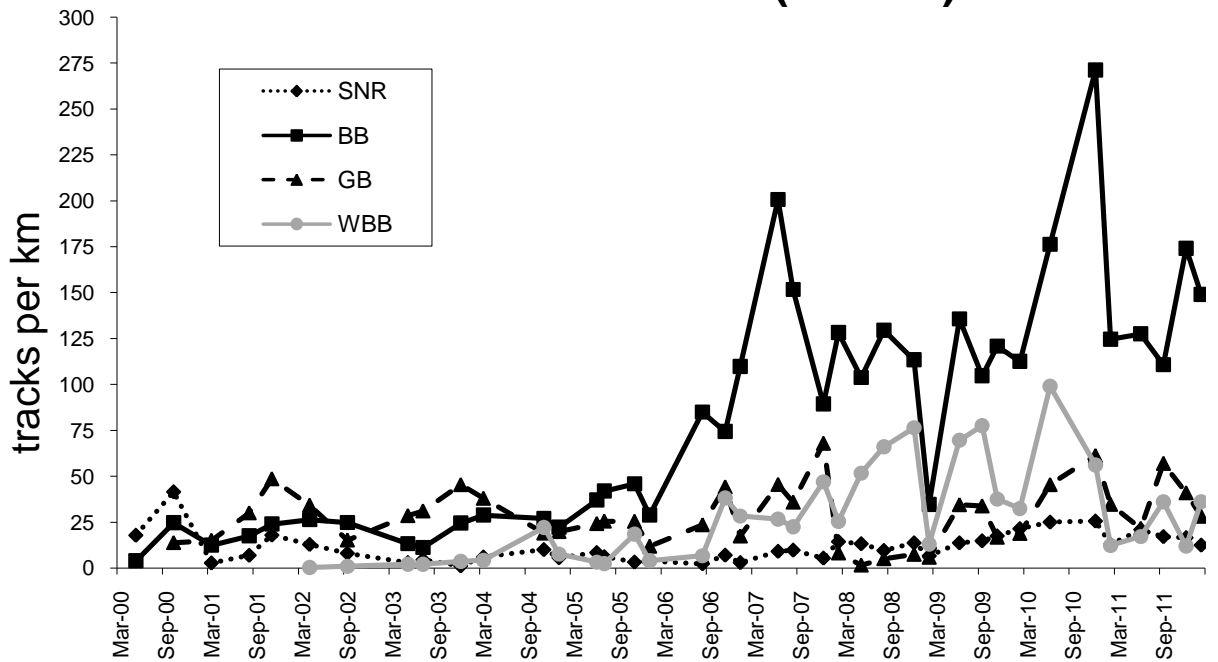


Fig. 3: Tracks per kilometre recorded on pooled transects within the Main Exclosure for each reintroduced species. Total length of transects is 10.6 km. SNR=stick-nest rat, BB=burrowing bettong, GB=greater bilby and WBB=western barred bandicoot.

### First Expansion (8 km<sup>2</sup>)

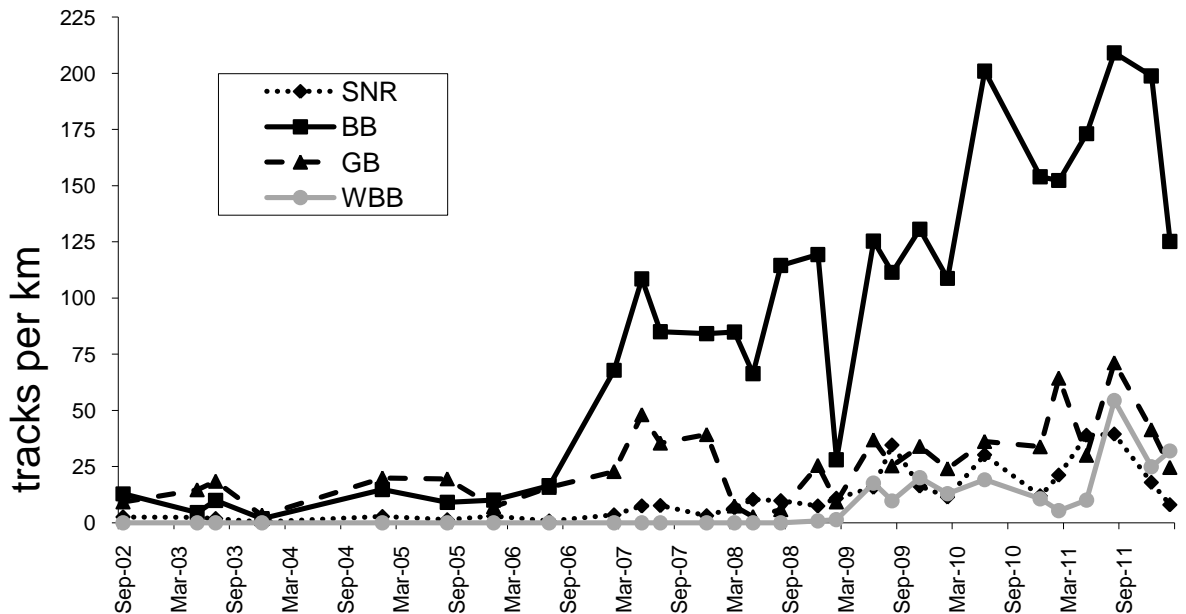


Fig.4: Tracks per kilometre recorded on pooled transects within the First Expansion for each reintroduced species. Total length of transects is 5.5 km. SNR=stick-nest rat, BB=burrowing bettong, GB=greater bilby and WBB=western barred bandicoot.

## Northern Expansion (30 km<sup>2</sup>)

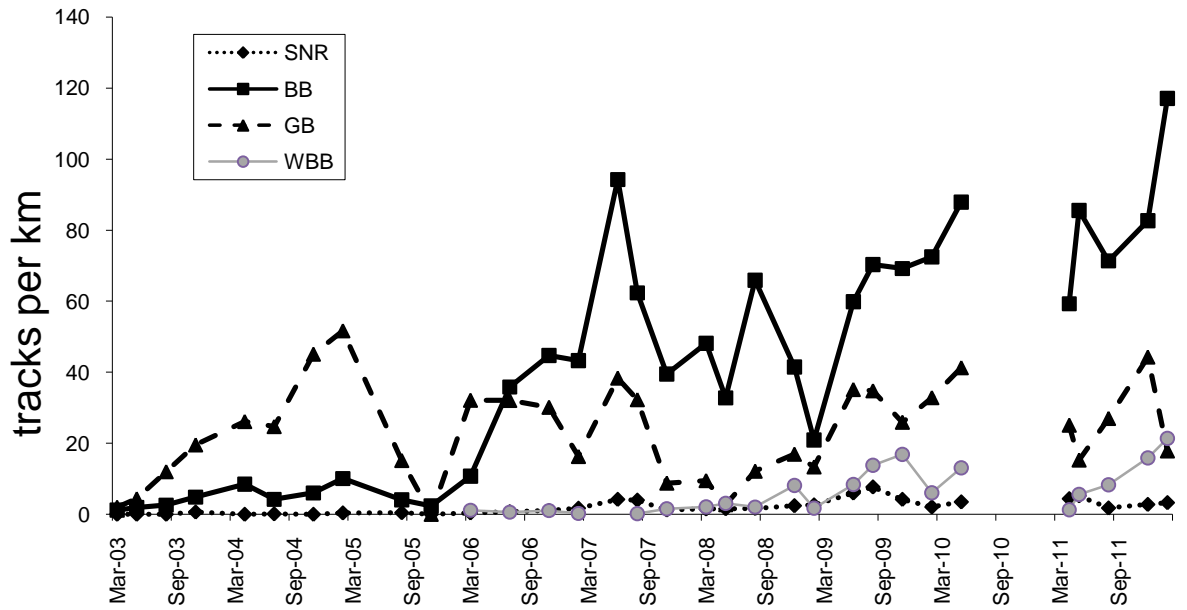


Fig. 5: Tracks per kilometre recorded on pooled transects within the Northern Expansion for each reintroduced species. No sampling was conducted between June 2010 and March 2011. Total length of transects is 14 km. SNR=stick-nest rat, BB=burrowing bettong, GB=greater bilby and WBB=western barred bandicoot.



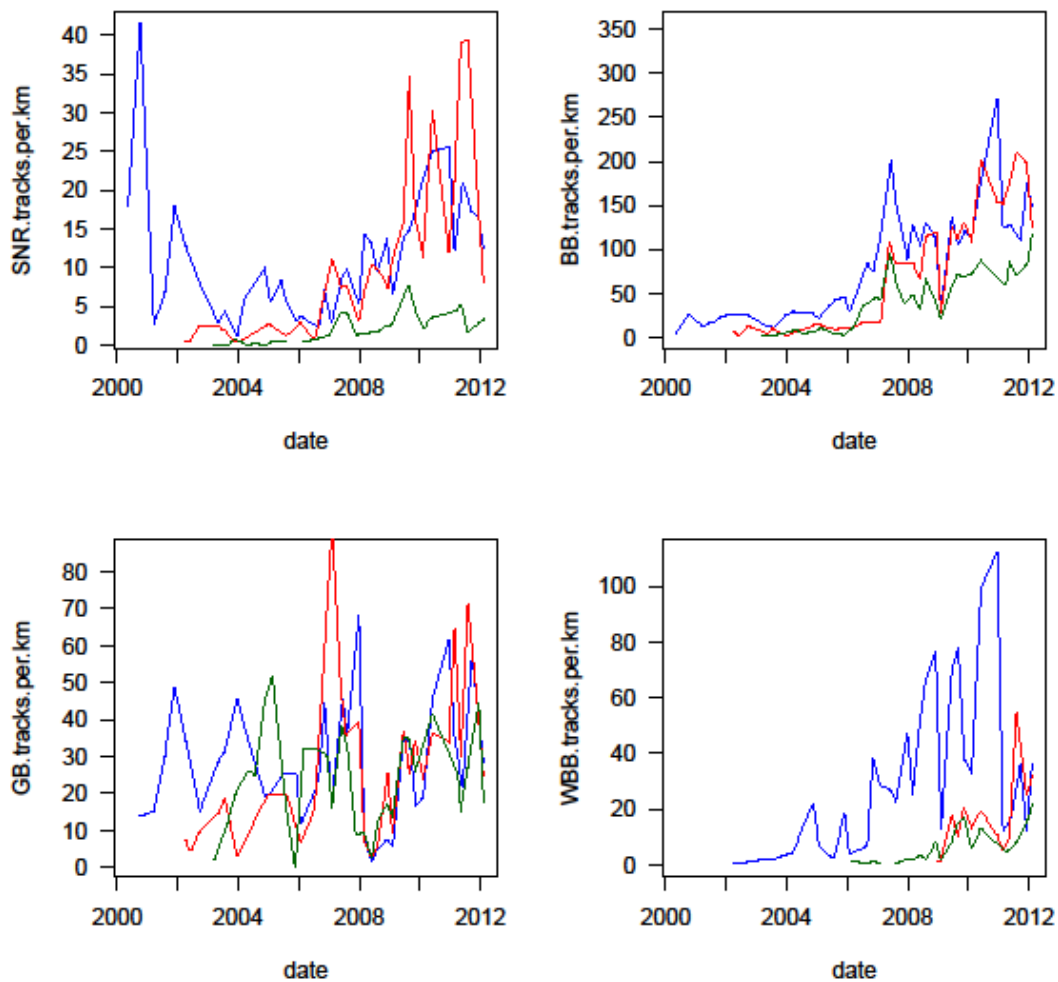


Fig. 6: Tracks per kilometre for each species in the three experimental exclosures. Blue=Main Exclosure, Green=First Expansion and Red=Northern Expansion. Note the different scales on the y-axis.

### 8.3.3 RATE OF INCREASE

The average rates of increase over the entire study period were calculated using the summary output from the highest-ranked model (Table 3). Within the Main Exclosure, the burrowing bettong and western barred bandicoot exhibited rates of increase over the entire study period that were more than double that of the other two species. The rates of increase in the Main Exclosure, where all species were first released, were significantly lower than the First Expansion. The western barred bandicoot was the only species to show a significant difference in the rate of increase between the First and Northern Expansion. When the founder sizes of each species in each exclosure were compared with rates of increase, there was no significant linear relationship ( $r^2=0.179$ ,  $F_1=3.39$ ,  $P=0.095$ ). The stick-nest rat had the largest founder size for two of the three exclosures but the lowest rate of increase in relation to time since release (Tables 1,3).

Table 3: The average rate of monthly increase over the study period for each species within the three exclosures. Rates are calculated from the coefficients of the slope of increase in track abundance over time since release). Stars indicate significant differences in rate of increase (slope) in relation to the First Expansion (\*\*\*=0.001, \*\*=0.01, \*=0.05).

Species	Rate of Increase		
	Main (14 km <sup>2</sup> )	First (8 km <sup>2</sup> )	North(30 km <sup>2</sup> )
burrowing bettong	0.02 (±0.002)***	0.04 (±0.003)	0.04 (±0.003)
western barred bandicoot	0.04 (±0.005)**	0.06 (±0.03)	0.05 (±0.01)*
greater stick-nest rat	0.006 (±0.003)***	0.03 (±0.004)	0.04 (±0.005)
greater bilby	0.001 (±0.003)*	0.01 (±0.004)	0.007 (±0.004)

Table 4: Approximate significance of smooth terms in the model. Variables that were present in the best model are indicated with an X. TSR=Time Since Release (months), IOD=Indian Ocean Dipole, Rain0-3= Rainfall recorded in the 3 months prior to sampling, Rain0-6 winter= winter rainfall recorded in the six month prior to sampling (in spring). Total Deviance = The total variance in track abundance explained by the entire model. TRS Deviance= The total variance in track abundance explained just by the variable ‘time since release’

Species	TSR	IOD	Temp	Rain 0-3	Rain 0-9	Rain 0-6 Winter	AIC <sub>c</sub>	df	Total Deviance	TSR Deviance
burrowing bettong	X	X	X	X			146.417	0	88.2%	80.8%
greater bilby	X	X	X		X		194.433	12	39.6%	18.0%
western barred bandicoot	X	X				X	156.341	8	78.5%	67.5%
greater stick-nest rat	X		X				152.000	10	76.9%	75.7%

### 8.3.4 EXPLANATORY VARIABLES

Exploratory analysis led to the removal of several variables from the models including longer rainfall lags of over 12 months, moonphase and month of sampling. The residual rainfall index was highly correlated with time since release and so was removed from the model. The remaining variables were included in the models and their contribution to the highest-ranking model varied according to species. A visual analysis of residual diagnostic plots for all four species suggested that the data were normally distributed and closely fitted the model. The terms in the highest-ranking model explained a large amount of the variance in the response variable (track counts) for the burrowing bettong (deviance=88.2%,  $r^2=0.868$ ), stick-nest rat (deviance=76.9%,  $r^2=0.752$ ) and western barred bandicoots (deviance=78.5,  $r^2=0.752$ ). The best model for the greater bilby explained the lowest amount of deviance (39.6%,  $r^2=0.326$ ) of the four species tested but the model was still considered a good fit.

The most significant variable influencing post-release track abundance was ‘time since release’ which explained between 18% and 81% of the total deviance in the monitoring data (Table 4). This variable was included in the highest-ranking model of all four species and comprised 92% of the total deviance for burrowing bettongs, 45% for bilbies, 86% for bandicoots and 98% for stick-nest rats. The highest-ranking model for the burrowing bettong also included the variables ‘Indian Ocean Dipole’ ( $F=_{1.87} 11.9$ ,  $P=<0.001$ ), ‘temperature’ ( $F=_{1.96} 4.6$ ,  $P=0.01$ ) and ‘rainfall in the three months prior to sampling’ ( $F=_{1.78} 1.63$ ,  $P=0.2$ ) (Table 1), cumulatively explaining the remaining 8% of the deviance in the model. Plots of the partial effects of the environmental covariates indicate that track abundance was strongly positively correlated with an increasing Indian Ocean Dipole (Fig.7). The relationship between track abundance and maximum temperature was weaker and non-linear, track abundance declined slightly when maximum temperatures on the day preceding sampling exceeded approximately 35°C (Fig.7). There was a weak association between track abundance and rainfall in the three months prior to sampling, with slightly higher track abundance recorded when rainfall in the preceding three months failed to reach 40mm. The delta and weight scores indicate that there was little difference in fit between the top three models. The third best model and associated plot (not presented here) suggested that total rainfall in the nine months prior to sampling was weakly positively correlated with track abundance.

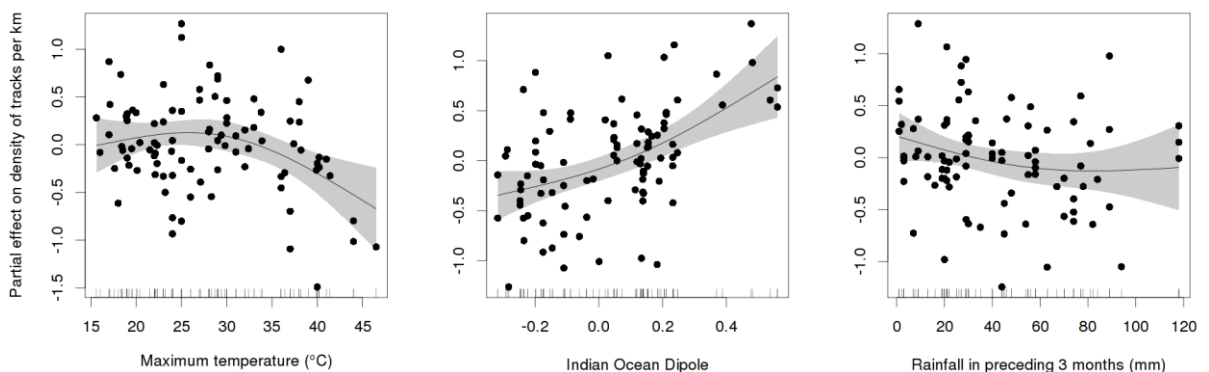


Fig. 7. Plots of the partial effects of environmental covariates on track abundance for the burrowing bettong. All enclosure data combined.

Unlike the other three species, the variable ‘time since release’ explained less than half the total deviance in the bilby model with three other variables ‘rainfall in the previous 9 months’ ( $F_{1,95}=9.03$ ,  $P<0.001$ ), ‘Indian Ocean Dipole’ ( $F_1=7.84$ ,  $P=0.006$ ), and ‘temperature’ ( $F_{1,96}=2.32$ ,  $P=0.1$ ) collectively explaining 21.6% of the deviance in the track abundance. There was a highly significant association between bilby track abundance and rainfall in the 9 months prior to sampling, track abundance declined when rainfall in the preceding 9 months dropped below approximately 100 mm (Fig.8). Bilby track counts declined from 68 to 1.6 per km in just five months during the worsening drought conditions in 2008. The positive relationship between track abundance and the Indian Ocean Dipole was not as strong as that recorded in the burrowing bettong but there was still a linear increase in track activity with increasing IOD (Fig. 8). There was a slight decline in track abundance when maximum temperatures on the day preceding sampling exceeded 40°C, but this association was weak.

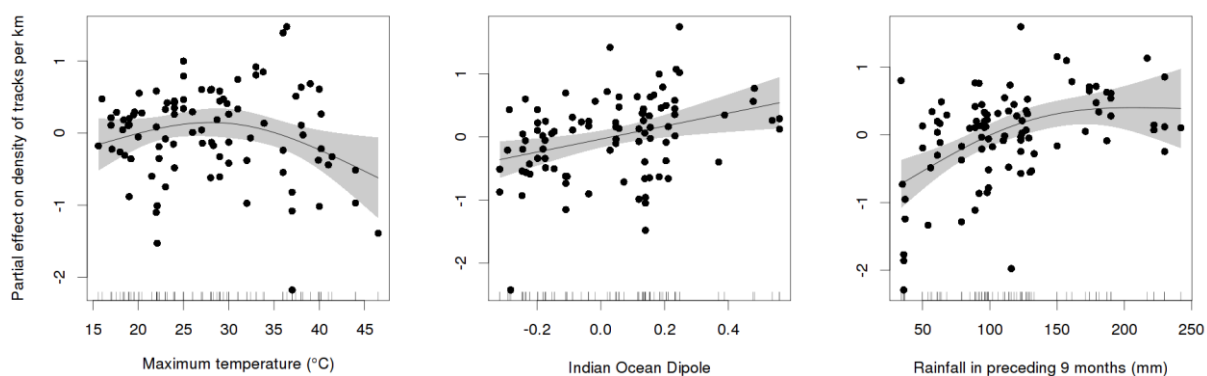


Fig. 8: Plots of the partial effects of environmental covariates on track abundance for the greater bilby. All enclosure data combined.

Time since release, the IOD ( $F_{1,98}=8.64$ ,  $P<0.001$ ) and winter rainfall in the 6 months prior to sampling ( $F_1=9.47$ ,  $P=0.003$ ) were environmental variables in the top ranked model explaining western barred bandicoot track abundance. There was a linear increase in bandicoot track abundance with increasing winter rainfall and a decline in abundance with low IOD in the 12 months prior to sampling (Fig. 9). Plots of the partial effects of environmental covariates indicate that bandicoot track abundance was lower when the IOD was negative.

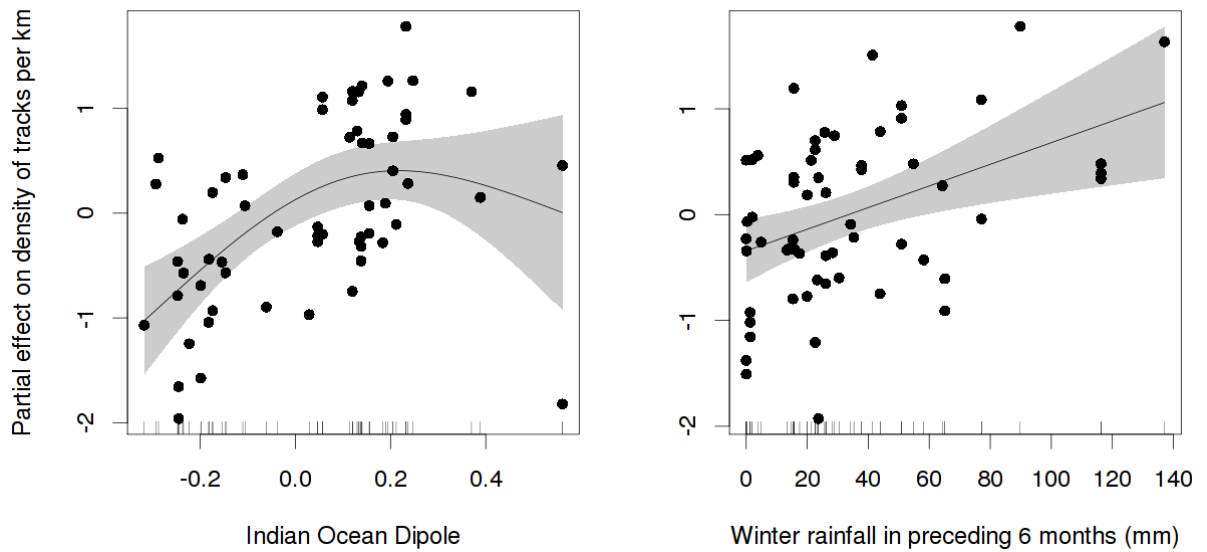


Fig. 9: Plots of the partial effects of environmental covariates on track abundance for the western barred bandicoot. All enclosure data combined.

Seventy five percent of the variation in greater stick-nest rat track abundance was explained in the model by the variable ‘time since release’, with only 1.2% explained by other environmental variables. There was a linear negative relationship between maximum temperature on the day preceding sampling and stick-nest rat track abundance ( $F_1=4.17$ ,  $P=0.04$ ) (Fig. 10). Breeding in stick-nest rats and western barred bandicoots at Arid Recovery generally occurs in the winter months (K. Moseby unpub. data) and these two species often exhibited cyclical patterns of track abundance with higher abundance recorded at the end of the breeding season in late winter and spring, and lower abundance in summer and early autumn (Figs. 3-6). Ninety rats were released into the Main Enclosure in 1999 but track abundance declined dramatically after the first summer (Fig. 6). When the proportional changes in track abundance over summer months were calculated for stick-nest rats in the Main Enclosure, abundance declined by 29-93% in 10 out of the 12 summer periods. When the summers with a proportional decline in track abundance were compared with the temperature variables tested, the decline was most strongly positively associated with average maximum summer temperature (Fig. 11,  $r^2=0.88$ ,  $P_1=58.45$ ,  $P<0.001$ ). The two summers where an increase in stick-nest rat track abundance was recorded were 2005/06 (15%) and 2009/10 (24%), the average maximum summer temperatures in these years was 37.0 and 35.8°C respectively compared to an average of 33-38°C in other years. Sampling periods occurred before the end of summer in both these instances.

There was a significant negative association between IOD and rainfall in the 12 months prior to sampling ( $r^2=0.352$ ,  $F_{31}=17.9$ ,  $P<0.001$ ).

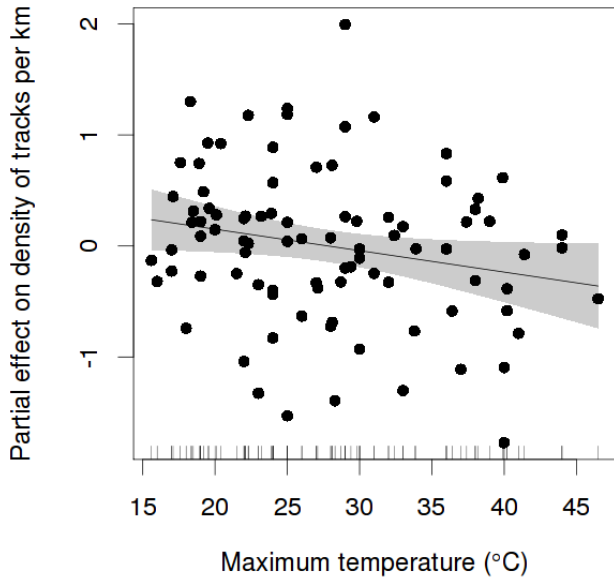


Fig. 10: Plot of the partial effects of temperature on track abundance for the greater stick-nest rat. All enclosure data combined.

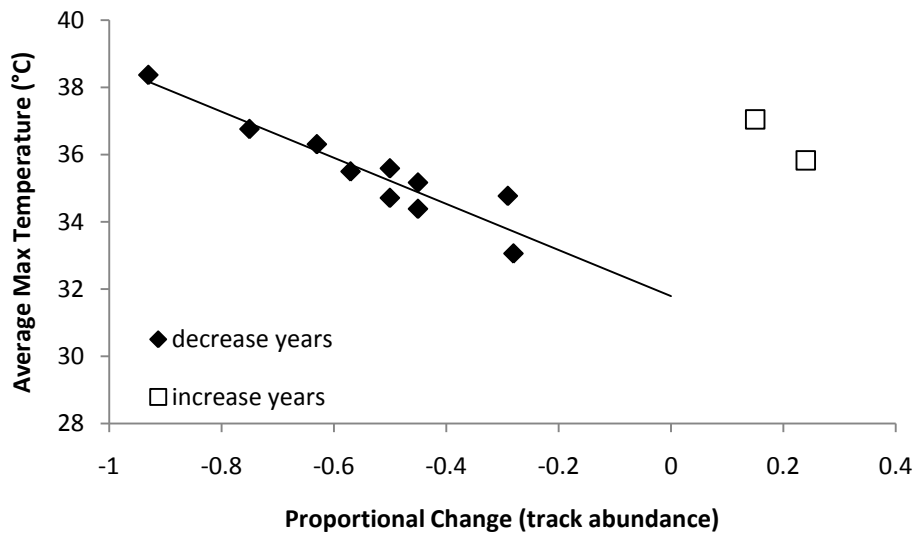


Fig. 11: Proportional change in stick-nest rat track abundance from the sampling period closest to the end of spring to the sampling period closest to the end of summer in relation to average maximum summer temperature. Only Main Enclosure data are shown. A linear regression line is fitted using only the years where a decline was recorded.

## 8.4 DISCUSSION

After more than 10 years of post-release monitoring, time since release was still the most significant variable influencing track abundance of the burrowing bettong, stick-nest rat and western barred bandicoot. Environmental variables had a significantly lower influence on track abundance in these species suggesting that post-release monitoring may need to continue for several decades before populations stabilise and the influences of environmental variables can be accurately ascertained.

Surprisingly, there was no relationship between founder size and the rate of increase since release suggesting that either the relative differences between founder sizes was inconsequential or more likely, that there were interspecific differences in breeding and survival unrelated to founder size. The higher rates of increase in the First Expansion and Northern Expansion relative to the original release site in the Main Enclosure may be partly due to stick-nest rats and bandicoots naturally dispersing over the Main Enclosure fence into subsequent release areas. The animals released into the Main Enclosure were from outside the study area and were likely to have taken longer to acclimatize to their new surroundings. Despite the greater stick-nest rat having the largest release size in two of the three enclosures, this species exhibited the slowest rate of increase after release. This is in direct contrast to stick-nest rats released onto Reevesby Island in South Australia where the population increased significantly for the first five years after release before undergoing a population crash after severe vegetation damage and subsequent nutritional stress. The population is now considered to be at carrying capacity and the vegetation has recovered to some extent (J. Van Weenen pers. comm.). The evidence of the negative influence of high summer temperatures on stick-nest rat abundance recorded in our study supports previous work at the study site where large numbers of radiocollared rats died during a heat wave (Moseby and Bolton 2000). Stick-nest rat reintroductions at arid zone sites are likely to experience lower rates of increase and possibly not reach the population extremes or starvation-induced crashes recorded in more mesic areas. The presence of native predators such as goannas and barn owls may also assist in regulating populations. Temperature increases predicted to occur under climate change may have major repercussions for arid zone stick-nest rat populations as even a 2°C increase in maximum summer temperature led to a 20% decline in stick-nest rat track abundance.

Abundance of stick-nest rat tracks was generally higher in spring than autumn and followed a cyclical pattern with abundance increasing after winter breeding events. This pattern is similar but opposite to that experienced by European hamsters (*Cricetus cricetus*) and the harvest mouse (*Micromys minutus*) which typically undergo a 65% and up to 95% decline over the winter months respectively (Weinhold 1998; Jordan 2003). Optimum release timing for the European hamster and harvest mouse has been suggested as spring coinciding with low population size and the commencement of population growth (Jordan 2003). Reintroductions are expensive and the value of each reintroduced individual can be maximised by timing releases when mortality is low and all animals have the potential to breed (Jordan 2003). We suggest that stick-nest rats should be released in autumn just prior to the breeding season which will both optimise breeding potential and allow rats maximum time to construct insulative nests before summer.

In the arid zone the greater bilby is capable of breeding in any month (McCracken 1983) and can produce up to four litters per year (Southgate *et al.* 2000). Bilbies have an accelerated life



history pattern (McCracken 1990) and are omnivorous and opportunistic dietary generalists (Gibson 2001, Bice and Moseby 2001), traits which enable them to rapidly colonise an area after good rainfall. These findings are supported by the post-release monitoring results which indicate that bilby activity is highly correlated with rainfall in the previous nine months. The relationship was non-linear with track abundance declining dramatically when rainfall in the previous 9 months dropped below 100 mm. The long term average rainfall at Roxby Downs is 166 mm per year (Read, 1995) equating to approximately 120 mm in a nine month period. Results from the present study suggest that rainfall deficiencies and drought may influence bilby abundance more so than large rainfall events. Gibson and Hume (2000) found that bilbies are susceptible to nutrient and water stress and have low fat stores, also suggesting they are vulnerable to drought conditions. Susceptibility to water stress may partly explain the lower track activity recorded during sampling periods when the maximum temperature preceding sampling exceeded 40°C. Hot evenings are likely to lead to reduced activity as bilbies attempt to conserve water and rest to stay cool.

Results have implications for management of reintroduced bilby populations. Reintroductions into single, small confined areas in the arid zone are likely to lead to severe genetic bottlenecks or local extinction during drought conditions due to the inability of bilbies to disperse or emigrate. Southgate and Possingham (1995) used Population Viability Analysis to suggest that extinction risk was lowered in reintroduced bilby populations by spatially segregating sub populations. Their reason was to ensure asynchronous episodes of drought and large rainfall events, two catastrophic events which significantly increased the probability of extinction. Large rainfall events were thought to increase extinction risk due to the associated increase in exotic predators responding to irruptions in local mammalian prey. Where exotic predators are excluded, such as the Arid Recovery Reserve, drought events would be the most likely cause of local extinction. In the arid zone, bilbies should be reintroduced to large exclosures or multiple unfenced sites to reduce the risk of genetic bottlenecks or population extinction during drought.

The association between higher winter rainfall and track abundance in western barred bandicoots may be related to breeding activity. Bandicoots do not breed continuously in the wild and have annual breeding cycles. Although breeding in mesic bandicoot species such as the eastern-barred bandicoot (*Perameles gunni*), southern brown bandicoot (*Isoodon obesulus*) and northern brown bandicoot (*I. macrourus*) is often correlated with rate of change of minimum temperature and daylength (Barnes and Gemmell 1984), rainfall is the main stimulus of reproductive activity in arid species such as the western barred bandicoot (Short *et al.* 1998). Richards and Short (2003) found the peak breeding season in western barred bandicoots to be June to September on Bernier and Dorre Islands but breeding extended into summer when there was above average spring and summer rainfall. Despite the higher, more predictable winter rainfall (70% of total annual rainfall, Short *et al.* 1998) on these islands compared with inland arid areas, western barred bandicoots released into acclimatisation pens at Arid Recovery also commenced breeding after the first significant post-summer rainfall event (K. Moseby pers. comm.). Winter and spring appear to be periods of higher breeding activity at Arid Recovery, with larger recruitment events occurring in years with higher winter rainfall.

Results suggest that western barred bandicoots should be reintroduced into arid areas immediately after significant autumn/winter rainfall. However, in arid zone areas where rainfall is unpredictable and aseasonal it may be difficult to time reintroductions to coincide with rainfall events and maximise population increase. Alternatively, it may be easier to avoid releasing animals during drought conditions which have been known to cause serious

significant population declines or even localised extinction in bandicoots. Western barred bandicoots declined significantly on Dorre Island during a prolonged drought from 1986 to 1989 (Short *et al.* 1998) and eastern barred bandicoots in Hamilton declined during a drought in 1966-68 (Seebeck 1990). A decline in western barred bandicoot tracks was also recorded at Arid Recovery during the drought in 2010.

The burrowing bettong exhibited the highest rate of increase over the 12 years after release and was the species least influenced by environmental variables. Macropods are commonly linked to problems of overabundance when predation pressure is low (Coulson and Eldridge 2010, Short 2009). Brush-tailed bettongs (*Bettongia lesueur*) exhibited exponential growth after release to Wedge Island (J. van Weenen pers.comm) and became overpopulated after reintroductions at Karrakamia Sanctuary (Short 2009). Burrowing bettongs at Scotia Sanctuary increased to high levels after release and were thought to impact on reintroductions of subsequent threatened species (Hayward *et al.* 2010). Overabundant populations of tamar wallabies and grey kangaroos reintroduced to islands have caused major grazing impacts (Copley 1994). These species often undergo a characteristic pattern of abundance after release which includes a latent establishment phase, an exponential increase phase, a significant decline or 'crash' phase followed by a more consistent and lower population level. The brush-tailed bettong population on Wedge Island eventually crashed due to starvation and has now stabilised at a much lower level. At Arid Recovery, burrowing bettongs have caused severe vegetation browsing and death of some plant species (K. Moseby pers. obs) suggesting that this species will also eventually undergo a starvation-induced population crash unless management actions are implemented. Intervention measures such as a predator reintroduction or bettong removal should be considered to protect *in situ* plant and animal species. The characteristic patterns of overabundance in many macropod species after release suggests these management interventions can be planned and should form part of the reintroduction strategy developed prior to release.

The positive relationship between the Indian Ocean Dipole and track abundance observed in three of the four species is perplexing considering that the IOD has been negatively correlated with winter rainfall in central Australia (Ashok *et al.*, 2003) as well as gecko recruitment and skink survival in our study region (Read *et al.* 2012). The IOD has also been negatively correlated with extreme rainfall events in southwest Western Australia (England *et al.*, 2006) and positively correlated with surface air temperatures in subtropical Australia during spring (Saji *et al.*, 2005). In our study IOD was not significantly correlated with time since release but a linear regression comparing rainfall in the 12 months prior to sampling and IOD revealed lower rainfall when IOD was high (as found by Ashok *et al.* 2003 and Read *et al.* 2012). However, the  $r^2$  value was small (0.352) suggesting that other environmental parameters were also influencing rainfall. The percentage of variance in track abundance of bettongs, bilbies and stick-nest rats explained by environmental variables in the best model was also generally very small suggesting this result may be an artefact of an association between IOD and another variable that was not measured. Additionally, the effect of smoothing IOD over a 12 month period preceding sampling may also have influenced results. Further research is required to understand the relationship between IOD and other environmental variables such as average temperature, humidity and rainfall patterns before this result can be explained.

Track abundance is related to activity and does not necessarily reflect true changes in abundance. Some results, particularly the non-linear relationships, were likely artefacts of the sampling method rather than real changes in abundance. Burrowing bettong and bilby track abundance was lower when maximum temperatures in the day preceding sampling exceeded

40°C. During these extreme weather events, animals may reduce nocturnal activity to conserve water and energy and spend more time resting to keep cool. When rainfall in the preceding 3 months failed to reach 40mm, bettong track activity also increased slightly, possibly related to increased foraging activity searching for food. Long term monitoring and multivariate analysis can assist in correct interpretation of monitoring results and provide recommendations for sampling methodology. For example, in keeping with the revised monitoring program advocated by Adaptive Monitoring (Lindenmayer and Likens 2009), Arid Recovery will now avoid sampling track transects on days after extreme temperature events.

## 8.5 CONCLUSIONS

Many ecologists have called for increased monitoring and research in reintroduction programs (Armstrong and McLean 1995, Sutherland 1998, Lindenmayer *et al.* 2012). Our study suggests that the influence of time since release on the abundance of reintroduced populations may extend beyond the 10 years post release monitoring advocated by Sutherland *et al.* 2010 for long-lived bird species. We suggest that post-release monitoring is conducted in three phases. The first phase should include intensive monitoring of individuals for a short period after release (e.g. radiotracking) to assess short term reintroduction outcomes. The second phase should include regular seasonal post-release monitoring of a population activity or abundance index until reintroduced population sizes are no longer influenced by time since release. Phase two seasonal monitoring, such as that conducted in this study, should continue until populations have completed several cycles of environmentally-induced fluctuations in order to accurately ascertain the influence of environmental variables on the fecundity, mortality and population trajectories of reintroduced species. Phase two will enable accurate assessment of medium term reintroduction outcomes as well as aid in the development of optimum future reintroduction protocols. The time frame required to reach phase two monitoring milestones may extend into several decades, particularly when release sites are large, founder populations small, and in desert environments where climatic patterns are unpredictable. Once populations have demonstrated their propensity to equilibrate to prevailing resource levels, the third phase of monitoring could involve a reduction in the frequency of monitoring to levels expected in natural wild populations. Monitoring should then be timed to key inflection points in the population cycle, for example when there is the greatest risk of extinction, or for cryptic species, when the probability of detection is highest.

## 8.6 ACKNOWLEDGEMENTS

This study was conducted by Arid Recovery, a conservation initiative supported by BHP Billiton, The South Australian Department for Environment and Natural Resources, The University of Adelaide and the local community. Funding was provided by a variety of sources including the Nature Foundation S.A., WWF Threatened Species Network, Western Mining Corporation, BHP Billiton, The Wildlife Conservation Fund and the Australian Federal Government's Natural Heritage Trust. We sincerely thank Brydie Hill for assistance with field work and all Arid Recovery staff who spent long hours clearing transects on quadbikes.

## 8.7 REFERENCES

- Armstrong, D. P. and McLean, I.G. (1995). New Zealand translocations: Theory and practice. *Pacific Conservation Biology* **2**, 39–54.
- Armstrong, D.P., Seddon P.J. (2008) Directions in reintroduction biology. *Trends in Ecology and Evolution* **23**, 20–25.
- Ashok, K., Guan, Z., Yamagata, T., 2003. Influence of the Indian Ocean Dipole on the Australian winter rainfall. *Geophysical Research Letters* **30**, 1821. doi:10.1029/2003GL017926
- Barnes, A. and Gemmell, R.T. (1984). Correlations between Breeding Activity in the Marsupial Bandicoots and some Environmental Variables. *Australian Journal of Zoology* **32**, 219-226.
- Bice, J.K. and Moseby, K.E. (2008). Diet of the re-introduced Greater Bilby (*Macrotis lagotis*; Peramelidae) and Burrowing Bettong (*Bettongia lesueur*; Potoroidae) in the Arid Recovery Reserve, Northern South Australia. *Australian Mammalogy* **30**, 1-12.
- Bolton, J. and Moseby, K.E. (2004). The activity of Sand Goannas *Varanus gouldii* and their interaction with reintroduced Greater Stick-nest Rats *Leporillus conditor*. *Pacific Conservation Biology* **101**, 193-201.
- Coulson, G. and Eldridge, D. (eds) (2010). *Macropods: The Biology of Kangaroos, Wallabies and Rat Kangaroos*. CSIRO Publishing, Collingwood, Victoria.
- Copley, P.B. (1994). *Translocations of native vertebrates in South Australia: a review*. In: *Reintroduction Biology of Australian and New Zealand Fauna*. (Ed. M. Serena) pp. 35-42. Surrey Beatty & Sons, Chipping Norton, NSW, Australia.
- Durbin, J. and Watson, G.S. (1971). Testing for Serial Correlation in Least Squares Regression III. *Biometrika* **58**, 1–19.
- England, M.H., Ummenhofer, C.C. and Santoso, A., (2006). Interannual rainfall extremes over southwest Western Australia linked to Indian Ocean climate variability. *Journal of Climatology* **19**, 1948-1969.
- Fischer, J. and Lindenmayer, D.B., 2000. An assessment of the published results of animal relocations. *Biological Conservation* **96**, 1-11.
- Gibson, L.A. (2001). Seasonal changes in diet, food availability and food preference of the greater bilby (*Macrotis lagotis*) in far south-western Queensland. *Wildlife Research* **28**, 121-134.
- Gibson, L. A. and Hume, I. D. (2000). Seasonal field energetics and water influx rates of the greater bilby (*Macrotis lagotis*). *Australian Journal of Zoology* **48**, 225-239.
- Griffith, B., Scott, J.M., Carpenter, J.W. and Reed, C. (1989). Translocation as a species conservation tool: status and strategy. *Science* **245**, 477-480.
- Hayward, M.W., Legge, S., Parsons, B., Page, M., Herman, K. And Mulder, E. (2010). Woylie re-introduction as part of the Australian Wildlife Conservancy's endangered species

- recovery program at Scotia Sanctuary, far western New South Wales, Australia. In: *Global Re-introduction Perspectives. Additional case-studies from around the globe.* (Ed. P.S. Soorae) pp202-207. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi. UAE, xii + 352.
- Jordan, M.J.R. (2003). Re-introduction protocols and considerations for small mammals. *Re-introduction NEWS* **22**, 39-43.
- Kemper, C. (1990). Status of bandicoots in South Australia. In: *Bandicoots and Bilbies.* (Eds. J. H. Seebeck, P. R. Brown, R. I. Wallis and C. M. Kemper.) pp. 67-72. Surrey Beatty & Sons, Sydney, Australia.
- Lindenmayer, D.B. and Likens, G.E. (2009). Adaptive monitoring: a new paradigm for long-term research and monitoring. *Trends in Ecology and Evolution* **24**, 482-6.
- Lindenmayer, D.B., Gibbons, P., Bourke, M., Burgman, M., Dickman, C.R., Ferrier, S., Fitzsimons, J., Freudenberger, D., Garnett, S.T., Groves, C., Hobbs, R.J., Kingsford, R.T., Krebs, C., Legge, S., Lowe, A.J., McLean, R., Montambault, J., Possingham, H., Radford, J., Robinson, D., Smallbone, L., Thomas, D., Varcoe, T., Vardon, M., Wardle, G., Woinarski, J. and Zenger, A. (2012). Improving biodiversity monitoring. *Austral Ecology* **37**, 285-294.
- McCracken (1983). Reproduction in the female bilby, *Mactois lagotis*. BSc. (Vet.) (Honours) Thesis:University of Sydney, New South Wales.
- McCracken (1990). Reproduction in the Greater Bilby, *Macrotis lagotis* (Reid)-a comparison with other perameloids. In: *Bandicoots and Bilbies* (Ed. J.H. Seebeck) pp 199-204.
- Morton, S.R. (1990). The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia* **16**, 201-213.
- Moseby, K.E. and Read, J.L. (2006). The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biological Conservation* **127**, 429-437.
- Moseby, K.E., Read, J.L., Paton, D.C., Copley, P., Hill, B.M. and Crisp, H.M. (2011). Predation determines the outcome of 11 reintroduction attempts in arid Australia. *Biological Conservation* **144**, 2863-2872.
- R Development Core Team (2012). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL <http://www.R-project.org/>.
- Read, J.L. (1995). Recruitment characteristics of the White Cypress Pine (*Callitris glaucophylla*) in arid South Australia. *The Rangeland Journal* **17**, 228-240.
- Read, J.L., Moseby, K.E., Briffa, J., Kilpatrick, A.D. and Freeman, A. (2011). Eradication of rabbits from landscape scale exclosures: pipedream or possibility? *Ecological Management and Restoration* **12**, 46-53.
- Read, J.L., Kovak, K.J., Brook, B.W. and Fordham, D.A. (2012). Booming during a bust; Asynchronous population responses of arid zone lizards to climatic variables. *Acta Oecologica* **40**, 51-61.

- Richards, J.D. and Short, J. (2003). Reintroduction and establishment of the western barred bandicoot *Perameles bougainville* (Marsupialia: Peramelidae) at Shark Bay, Western Australia. *Biological Conservation* **109**, 181-195.
- Saji, N.H., Goswami, B.N., Vinayachandran, P.N., Yamagata, T. (1999). A dipole mode in the tropical Indian Ocean. *Nature* **401**, 360-362.
- Saji, N.H., Ambrizzi, T. and Ferraz, S.E.T. (2005). Indian Ocean Dipole mode events and austral surface air temperature anomalies. *Dynamic Atmospheric Oceans* **39**, 87-101.
- Seddon, P. J., Armstrong, D. P. and Maloney R. F. (2007). Developing the science of reintroduction biology. *Conservation Biology* **21**, 303-312.
- Short, J. (2009). *The characteristics and success of vertebrate translocations within Australia*. Australian Government Department of Agriculture, Fisheries and Forestry, Canberra.
- Short J., Richards J.D. and Turner B. (1998). Ecology of the Western Barred Bandicoot (*Perameles bougainville*) (marsupialia: Peramelidae) on Dorre and Bernier Islands, Western Australia. *Wildlife Research* **25**, 567-86.
- Soorae, P.S. (Ed) (2010). *Global Re-introduction Perspectives. Additional case-studies from around the globe*. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi. UAE, xii + 352.
- Southgate, R.T., Christie, P. and Bellchambers, K (2000). Breeding biology of captive, reintroduced and wild greater bilbies *Macrotis lagotis* (Marsupialia:Peramelidae). *Wildlife Research* **27**,621-628.
- Southgate, R. and Possingham, H. (1995). Modelling the reintroduction of the greater bilby *Macrotis lagotis* using the metapopulation model Analysis of the Likelihood of Extinction. *Biological Conservation* **73**, 151-160.
- Sugiura, N. (1978). Further analysis of the data by Akaike's information criterion and the finite corrections. *Communications in Statistics, Theory and Methods* **A7**, 13-26.
- Sutherland, W. J. (ed.) (1998). *Conservation Science and Action*.
- Blackwell Science, Oxford.Sutherland, W.J., Armstrong, D., Butchart, S.H.M., Earnhardt,J.M., Ewen, J. Jamieson, I., Jones, C.G., Lee, R., Newbery, P., Nichols, J.D., Parker, K.A., Sarrazin, F., Seddon, P.J., Shah, N. and Tatayah, V. (2010). Standards for documenting and monitoring bird reintroduction projects. *Conservation Letters* **3**, 229-235.
- Weinhold, U. (1998). Bau und Individuendichte des Feldhamsters (*Cricetus cricetus* L., 1758) auf intensiv genutzten landwirtschaftlichen Flächen in Nordbaden: Ökologie und Schutz des Feldhamsters. Halle/Saale: 277-288.
- Wood, S.N. (2004). Stable and efficient multiple smoothing parameter estimation for generalized additive models. *Journal of the American Statistical Association* **99**, 673-686.
- Wood, S.N. (2006). *Generalized Additive Models: an introduction with R*, CRC.
- Wood, S.N. (2011). Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. *Journal of the Royal Statistical Society (B)* **73**, 3-36.

## 9. CHAPTER NINE: CONCLUSIONS

This chapter is presented in four sections. Section one synthesises results from chapters two to eight and outlines a number of guiding principles for improving mammal reintroduction in the Australian arid zone. In the second section I discuss some of the limitations of my research program. The third section builds upon results of the first two sections to outline directions for future research and finally my outcomes are discussed in relation to a number of mammal decline paradigms relevant to arid Australia.

### 9.1 GUIDING PRINCIPLES

#### **Excluding cats and foxes will significantly increase the likelihood of mammalian reintroduction success.**

Predation was the single largest factor influencing reintroduction success in my study. The successful reintroductions within the Arid Recovery Reserve where cats, *Felis catus*, and red foxes, *Vulpes vulpes*, were excluded are consistent with results found in other exclosures and islands such as Warrawong Sanctuary (John Wamsley pers.comm.), Scotia Sanctuary (Matt Hayward pers. comm.), Heirisson Prong (Richards and Short 2003), Western Australian islands (Morris 2000), Escape Island (Moro 2003) and Thistle Island (S.A. Dept for Environment and Natural Resources). Clearly, when exotic mammalian predators are excluded, successful reintroductions of many critical-weight range mammals are almost assured. The exception to this rule is when the release animal is towards the lower end of the critical weight range or (semi) diurnal and susceptible to other predator taxa such as raptors or reptiles (e.g. numbats, Moseby *et al.* 2011, stick-nest rats in Yookamurra, S.A. Dept for Environment and Natural Resources, unpublished data).

#### **Current cat control techniques are insufficient to enable widespread reintroductions into arid zone areas for sensitive species.**

Dickman (1996) considered species up to 2 kg to be affected by cat predation but juveniles of species weighing as much as 4.5 kg could also be sensitive. Read and Ward (2011) consider cat predation on juveniles to be the primary cause of continued suppression of 4 kg warru (black-flanked rock wallaby, *Petrogale lateralis*) populations in northern South Australia. Adult bilbies which can weigh as much as 2.5 kg, exhibited high survival after release outside the Arid Recovery Reserve but the population became extinct after cat predation on juveniles. Arid zone mammal species that fall below the 2 kg weight limit include the burrowing bettong and mala. Species weighing less than 220g are considered most vulnerable to cat predation (Dickman 1996) including the stick-nest rat, western barred bandicoot and numbat. Although some conservation programs claim to have successfully reintroduced some of the less sensitive species into arid areas where cats and foxes are controlled (see bilbies at Project Eden and Mt Gibson in Western Australia, DEC unpubl. data), long term success has not been proven and fencing is still used to limit the impacts of predation. Proponents such as the Australia Wildlife Conservancy, Western Australian Department for Environment and

Conservation, N.S.W. National Parks and Wildlife Service and Arid Recovery have attempted releases into large unbounded arid or semi-arid sites but to date long term re-establishment has not been achieved (Christensen and Burrows 1994, M. Hayward pers. comm. Australian Wildlife Conservancy, Priddel and Wheeler 2002; Moseby *et al.* 2011). No successful releases of mammal species highly sensitive to cat predation have occurred in arid areas without the use of exclusion fencing. Aerial baiting, shooting, ground baiting and trapping can all reduce cat and fox abundance but sustaining low levels of cat abundance is difficult due to their reluctance to ingest baits. It is unlikely that arid zone mammal species less than 2 kg or with independent juveniles susceptible to cat predation can currently be successfully reintroduced into unfenced arid zone areas.

### **Reintroductions of arid zone species into unfenced release sites should target species that are not sensitive to cat predation.**

Low levels of fox abundance could be sustained through quarterly aerial baiting of 10 baits per km<sup>2</sup> at my study site. Results are encouraging and support studies in mesic areas which have also found regular baiting to be successful at controlling foxes (Saunders *et al.* 1995). The home ranges and movement of arid zone foxes are much larger than their mesic counterparts (Moseby *et al.* 2009b) and so large baited areas of 1800 km<sup>2</sup> and 40 km in width were required to reduce invasion. Successful reintroductions of species that are less susceptible to cat predation may be able to occur in the arid zone through the use of aerial baiting. A successful reintroduction of a captive-bred yellow footed rock wallaby, *Petrogale xanthopus*, population in the Flinders Ranges occurred when fox control was initiated (Andrews *et al.* 2010) and a population of tammar wallabies (*Macropus eugenii*) reintroduced to an area subject to intensive fox control in South Australia is currently extant (Sharp *et al.* 2010). Species such as the black flanked rock wallabies and western quoll may be suitable for reintroduction to arid areas subject to intensive fox control due to their larger size (former) or more aggressive behavior (latter). Ideally fox control techniques would not target dingoes to enable them to exert pressure on foxes and cats but currently this option is not available. My research found that rabbit abundance increased after aerial baiting possibly due to the reduction in predation pressure from foxes. Implementing rabbit control in aerial baited areas may be required to prevent rabbits from reducing food or vegetation cover for reintroduced species and could also improve predator baiting outcomes.

### **Soft releases should occur in situations of high predation risk, unbounded released sites, supplementation or for social or sedentary species**

My results, and those from other reintroduction attempts, suggest that when predation risk is low and the release site is contained (islands or fenced mainland sites) hard releases are likely to be successful and more cost effective. The increased cost associated with soft release strategies suggests that they should be restricted to those circumstances where they are likely to be beneficial. When predation risk is high or the release site is unbounded, a soft release may retain animals close to the release site where effective predator control can be maintained. When social, sedentary species that invest heavily in their shelter sites are being reintroduced, soft releases should be considered on ethical grounds, even in situations of low predation risk. For these species, soft releases may prevent panic dispersal, minimize stress and give them time to form social groups. Where genetic supplementation is required into an



already dense population, a soft release strategy may ensure new genetics are successfully absorbed into the population.

**Wild-caught founders should preferentially be used in reintroductions but if predation pressure is low then releases of captive stock are likely to be successful.**

Species-specific predation pressure should be carefully assessed prior to development of the release strategy. If releases are occurring into areas where predation pressure is moderate or high (including any situations where cats and foxes are present even in low numbers), then wild stock should be used. When exotic predators are excluded, captive stock may be used but differences in post-release survival and behaviour are likely to be evident in some species, particularly smaller species that are susceptible to aerial or reptilian predators. I concur with Snyder *et al.* (1996) that captive-bred species with higher levels of parental care may exhibit higher mortality or behavioural changes than those with precocious young.

**Conducting long-term post-release monitoring can improve future reintroduction strategies and contribute to knowledge of threatened species ecology and decline.**

Long-term post-release monitoring is critical for understanding the ecological role of locally-extinct species and for refining future release strategies. Monitoring can assist with optimising the timing of future releases, size of release area and the order in which species are reintroduced. Monitoring results can also be used to plan future management strategies for over-population, dispersal and drought responses. My research suggests that populations of reintroduced species may not stabilise for decades and that long term monitoring is required to understand population dynamics and accurately assess reintroduction outcomes. Three phases of post-release monitoring were outlined: Phase One - intensive monitoring of individuals immediately after release, Phase Two - intensive population monitoring until the influence of time since release on population abundance is not significant and environmental drivers have been identified and Phase Three – reduced intensity of population monitoring to levels conducted on wild, extant populations. Long-term post-release monitoring should be considered best practice for reintroduction programs and preferably include measures of other biota that may be impacted on by the reintroduced species (e.g. plants, other *in situ* fauna).

## 9.2 LIMITATIONS OF MY RESEARCH

### **Pseudoreplication**

My study was conducted at a single location (the Arid Recovery Reserve) and as such was subject to problems of pseudoreplication. Where possible this was limited by using temporal replication or by making use of the different exclosures within the Arid Recovery Reserve. Additionally, I have improved replication by comparing my results with other similar predator-free exclosures in Australia. In general, my results support those found in other arid zone sites, which considerably strengthened my research outcomes.

### **Low sample size**

One of the difficulties in working with threatened species is the low numbers available for experimentation. Sample sizes for releases were generally small, in some instances hampering the use of appropriate statistical analyses and my ability to generate clear conclusions. In hindsight, more replication and fewer treatments in some sections may have yielded more robust results.

### **Logistical difficulties**

Working at a remote and large scale arid zone field site presented logistical challenges that could not always be overcome. Locating radiocollared animals after release, particularly outside the Arid Recovery Reserve, was problematic and often affected results. Long hours were spent searching for animals after release on foot or quadbike and results were not an accurate reflection of effort. Significant travel time and kilometres were expended capturing cats and foxes over the large geographical area required to ensure independence of baiting treatments.

## **9.3 DIRECTIONS FOR FURTHER RESEARCH**

### **The development of broadscale cat control methods suitable for arid zone conditions**

My research indicates that whilst foxes can be controlled through aerial baiting, broadscale control of cats using poison baits is unlikely to be successful or sustained in the long term. Despite some practitioners reporting success with aerial baiting at times when alternative prey is low (Algar and Burrows 2004), most researchers conclude that broadscale cat control is problematic (Risbey *et al.* 1997; Denny and Dickman 2010). New technologies need to be developed that target cats and are suitable for arid zone conditions. Arid zone conditions are extremely harsh and any new control method would need to withstand summer temperatures of up to 50°C (in the shade) and winter temperatures that drop below freezing. Droughts and windstorms can raise high dust loads, damaging sensitive equipment. Remoteness means that control is also logistically difficult and expensive. On a positive note, high insolation rates make the arid zone suitable for solar power which may enable automotive poison delivery devices to be powered for long periods.

One new control method currently in the trial stage is an automotive poison delivery device that relies on poison ingestion through grooming rather than hunger (Read 2010). Initial results are promising but the method still relies on placing individual control stations out at sufficient density to control cats. This may be difficult over large areas and will require significant logistical and funding resources to maintain in the long term.

Ideally, a biological control agent would be developed for the feral cat with the potential for owners to immunize their domestic cats against the agent. Such a control strategy would enable broadscale implementation and widespread threatened species recovery. Although diseases and viruses are known to become less successful over time as target species develop tolerance, the impact of myxomatosis and calicivirus on reducing rabbit abundance in arid Australia is a good example of what can be achieved. Large scale recovery of many extant

threatened plant and animal species in the South Australian arid zone has been attributed to the reduction in rabbits both through a response in seed producing plants as well as lowering the abundance of predators that feed on rabbits. Major obstacles to developing a biocontrol agent for cats are two-fold. First, public support is minimal due to the high number of households with pet cats and/or naïve ethical opposition to cat control. The lack of understanding of the damage inflicted on our native wildlife by feral cats prevents large scale support for cat control. Secondly, cats do not cause damage to agriculture or mining which limits industry support (both financial and philosophical) for development of control methods.

### **The development of predator thresholds**

Little attention has been afforded to the value of developing thresholds of predator activity in Australian reintroduction biology. New Zealand studies have highlighted the importance of identifying predator thresholds for guiding and triggering management decisions (Armstrong *et al.* 2006) but Australian practitioners continue to attempt to reintroduce threatened species into areas where exotic predators are present. There is often little attempt to measure predator activity and set targets for future reintroduction attempts. A lack of standardized data collection is also an issue whereby results cannot be compared between release sites across Australia. Future research is required to establish agreed monitoring protocols and methods for monitoring predators as well as reintroduced populations. Reintroductions should then be used as experiments to test predator thresholds. For example, at Arid Recovery releasing bilbies into areas with 20% cat presence on transects was unsuccessful therefore future releases should be trialed at 10 or 15%. Predator thresholds are also likely to be influenced by the abundance of alternative prey and the habitat quality including the availability of food and shelter. These factors should also be measured and used as covariates in predictive models of predator thresholds.

### **The use of dingoes to improve reintroduction success**

The majority of arid zone threatened species which are still extant in South Australia are located north of the dingo fence where dingoes are present (Biological Survey Database of South Australia). Several studies have recorded lower fox abundance north of the fence (Newsome *et al.* 2001; Letnic *et al.* 2009) and have attributed the survival of many threatened species in the arid zone to the presence of dingoes (Letnic *et al.* 2009). However, dingoes may be providing benefits to *in situ* threatened species but their ability to increase reintroduction success of locally-extinct species needs to be tested experimentally. It is likely that dingoes will not benefit all species or not suppress cats and foxes in all habitats. I suggest that dingoes will have their greatest influence in open, sparsely-vegetated arid habitats and with cryptic threatened species. Threatened species most likely remain extant because they fit this profile, suggesting that many locally-extinct threatened species may not benefit from the presence of dingoes during reintroduction. In fact, surplus killing by dingoes immediately after release as recorded in the burrowing bettong reintroduction at Arid Recovery may contribute to reintroduction failure. My study supports the theory that dingoes can suppress cat and fox abundance but further work is required to determine if that translates to a net benefit for reintroductions of different species under various conditions. The experimental reintroduction of threatened species into areas where dingoes are present could be conducted but ideally dingoes would be introduced into an area after a population of

reintroduced threatened species was already established to prevent panic dispersal and surplus killing. Soft releases may need to be used to ensure initial establishment.

#### 9.4 ARID ZONE MAMMAL DECLINE

There are many theories surrounding the cause of mammal decline in arid Australia including predation, pastoralism, rabbits, disease, hunting, and changes to fire regimes (Jones 1924, Finlayson 1961, Newsome 1971, Burbidge and Fuller 1979, Burbidge and McKenzie 1989, Kemper 1990, Short 1998). My research suggests that predation from introduced cats and foxes may have been a major factor responsible for mammal decline in arid Australia. The vast majority of mammal reintroductions to a cat and fox free enclosure succeeded whilst reintroductions in the same habitat in the presence of even low abundance of cats and foxes failed due to predation. Despite a severe drought event, reintroduced species continued to survive and thrive within the Arid Recovery enclosure. The large number of other successful reintroductions to cat and fox-free areas including some with significantly altered habitats and even the continued presence of rabbits, supports the theory that predation was possibly the most significant factor responsible for arid zone mammal extinctions. There was also some evidence to support the theory of rabbits contributing to arid zone mammal decline through hyperpredation (Smith and Quin 1996). Rabbits are common at Roxby Downs and feature prominently in the diet of cats and foxes (Read and Bowen 2001). When rabbit abundance declined after the introduction of calicivirus in 1996, fox and cat abundance also declined (Read and Bowen 2001) but even a low rabbit population was enough to support a resident population of feral cats and foxes that preyed on reintroduced species after release. Rabbit abundance increased after aerial baiting of cats and foxes, further reducing vegetation cover for native species and hindering attempts to bait feral cats which prefer to feed on live prey when available.

The theory of refugia outlined by Morton (1991) was not supported by my study as I found threatened species could survive in low abundance throughout the Reserve during drought rather than contracting back to certain areas with higher productivity. Results from my dingo study support the theory of mesopredator release (Crooks & Soulé 1999) as dingoes were found to eradicate foxes from the dingo pen and reduce cat abundance. The intensive control of dingoes that occurred in pastoral areas last century is likely to have accelerated the decline of arid zone mammal species by allowing cats and foxes to increase in abundance.

Exotic predators and introduced rabbits are now well established in arid Australia. Rabbits provide a reliable food source for cats and foxes enabling them to persist in areas even during drought conditions. A search of almost any dune system in the South Australian arid zone will reveal that cats, foxes and rabbits are ubiquitous. Calicivirus has helped reduce rabbit, cat and fox abundance but even low levels of predation are enough to cause reintroduction failure in small to medium arid zone mammals. Until a cost-effective broadscale method of cat control is developed, reintroductions of small to medium sized threatened species are likely to succeed only behind wire fences. Widespread restoration of our arid zone mammal fauna is currently an aspirational target rather than an imminent possibility.

## 10. ACKNOWLEDGEMENTS

Starting a PhD three months before the birth of your third child, when you are fast approaching 40 and moving to a remote outpost on the Eyre Peninsula could be considered the start of a mid life crisis. However, I have relished the challenge and thoroughly enjoyed writing this PhD in between nappy changes, school bus runs and solar power outages. I hope that the resulting publications will assist practitioners when designing reintroduction programs in arid Australia.

First and foremost, I would like to thank the founders of Arid Recovery for having the vision and perserverence to restore a small patch of the Australian arid zone and enable vital research to be conducted on arid zone threatened species. Three of these founders, namely John Read, David Paton and Peter Copley have also provided thoughtful insights into my PhD topic and assisted in the research direction taken. I am indebted to the significant number of Arid Recovery staff, students and volunteers who have cheerfully and determinedly assisted with field work during this study, despite the often extreme weather conditions. I would particularly like to thank John Read, Brydie Hill, Helen Crisp, Andrew Freeman, Jackie Bice, Adam Bester, Nicki Munro, Jenny Stott, Bree Galbraith, Adam Kilpatrick, Heather Neilly, Pete Paisley, Amber Cameron and Tyrone Lavery for their enthusiasm and input. We all share the same dream; an arid zone without exotic rabbits, foxes and cats that is teeming with native species. Let's hope we live to see it a reality.

My family has been extremely tolerant of the long hours I have spent on the computer, particularly my two year old Jarrah who has trouble pronouncing "PhD" let alone understanding what one is. Thanks to my partner John for reading draft manuscripts at short notice and always questioning my theories and ideas. This thesis has been considerably improved through his role as devil's advocate! My mum graciously assisted with proof reading earlier drafts. Finally, thanks to my supervisor David Paton who tolerated long lapses of silence interspersed with frantic email requests for assistance, and always helped whenever help was needed.

## 11. BIBLIOGRAPHY

- Abbott, I. (2000). Improving the conservation of threatened and rare mammal species through translocation to islands: case study Western Australia. *Biological Conservation* **93**, 195-201.
- Abbott, I. (2002). Origin and spread of the cat, *Felis catus*, on mainland Australia, with a discussion of the magnitude of its early impact on native fauna. *Wildlife Research* **29**, 51-74.
- Algar, D.A., Angus, G.J., Fuller, P.J. and Onus, M.L. (2002a). Operation Wanjarri: optimising feral cat baiting strategies in the arid zone. Report to Environment Australia. Department of Conservation and Land Management, Western Australia.
- Algar, D., Angus, J.G., Williams, M.R. and Mellican, A.E. (2007). Influence of bait type, weather and prey abundance on bait uptake by feral cats (*Felis catus*) on Peron Peninsula, Western Australia. *Conservation Science Western Australia*. **6**, 109-149.
- Algar, D. and Brazell, R.I. (2008). A bait-suspension device for the control of feral cats. *Wildlife Research* **35**, 471-476.
- Algar, D.A., Burbidge, A.A. and Angus, G.J. (2002b). Cat eradication on Hermite Island, Montebello Islands, Western Australia. In: *Turning the Tide: Proceedings of the international conference on the eradication of island invasives*. (Eds. C. R. Veitch and M. N. Clout.) pp. 14-18. IUCN: SSC Invasive Species Specialist Group: Gland, Switzerland and Cambridge, UK.
- Algar, D. and Burrows, N.D. (2004). Feral cat control research: *Western Shield* review-February 2003. *Conservation Science Western Australia* **5**, 131-163.
- Algar, D. and Sinagra, J.A. (1996). *Methods of broadscale control of feral cats in Western Australia*. Unpublished report to the Australian Nature Conservation Agency, Feral Pests Program Project No. 11. ANCA, Canberra.
- Algar, D. and Smith, R. (1998). Approaching Eden. *Landscape*, **13**, 28-34.
- Andrews, L., Bigwood, S. and Barlow, S. (2010). The re-introduction of the yellow-footed rock-wallaby to the northern Flinders Ranges, South Australia. In: *Global reintroduction perspectives: Additional case studies from around the globe*. (Ed. P. Soorae) pp 215-219. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi, UAE.
- Anthony, L.L. and D.T. Blumstein, T.D., (2000). Integrating behaviour into wildlife conservation: the multiple ways that behaviour can reduce  $N_e$ . *Biological Conservation* **95**, 303-315.
- Arid Recovery (2008). *Arid Recovery Annual Report*. Roxby Downs, South Australia.
- Armstrong, D. P. and McLean, I.G. (1995). New Zealand translocations: Theory and practice. *Pacific Conservation Biology* **2**, 39-54.
- Armstrong, D., Raeburn, E.H., Lewis, R.M. and Ravine, D. (2006). Modelling vital rates of a New Zealand robin population as a function of predator control. *The Journal of Wildlife Management* **70**, 1028-1036.

Armstrong, D.P. and Seddon P.J. (2008). Directions in reintroduction biology. *Trends Ecology and Evolution* **23**, 20–25.

Ashok, K., Guan, Z., Yamagata, T., 2003. Influence of the Indian Ocean Dipole on the Australian winter rainfall. *Geophysical Research Letters* **30**, 1821. doi:10.1029/2003GL017926

Australian Bureau of Meteorology (2011). Australian Bureau of Meteorology Website. <http://www.bom.gov.au>

Backhouse, G.N., Clark, T.W., Reading, R.P. (1994). Reintroductions for recovery of the eastern barred bandicoot *Perameles gunnii* in Victoria, Australia. In: *Reintroduction Biology of Australian and New Zealand Fauna*. (Ed. M. Serena) pp 209-218. Surrey Beatty and Sons, Chipping Norton, NSW.

Bajomi, B. (2010). *Reintroduction of endangered animal species: complementing the IUCN guidelines*. Review protocol. <http://www.environmentalevidence.org/Documents/Draftprotocol86.pdf>. (accessed the 16th February 2011). Centre for Environmental Science, Eötvös Loránd University of Sciences, Budapest, Hungary.

Banks, P.B. (1998). Responses of Australian bush rats, *Rattus fuscipes*, to the odor of introduced *Vulpes vulpes*. *Journal of Mammalogy* **79**, 1260–1264

Banks, P.B. and Dickman, C.R. (2007). Alien predation and the effects of multiple levels of prey naivete'. *Trends in Ecology and Evolution* **22**, 229–230

Banks, P.B., Norrdahl, K. and Korpimäki, E. (2002). Mobility decisions and the predation risks of reintroduction. *Biological Conservation* **103**, 133-138.

Barnes, A. and Gemmell, R.T. (1984). Correlations between Breeding Activity in the Marsupial Bandicoots and some Environmental Variables. *Australian Journal of Zoology* **32**, 219-226.

Beck, B. B., Rapaport L. G., Stanley-Price, M. R., and Wilson, A. C., (1994). Reintroduction of captive-born animals. In: *Creative conservation: interactive management of wild and captive animals*. (Eds. P.J.S. Olney, G.M. Mace and A.T.C. Feister) pp 265-286. Chapman & Hall, London.

Bester, A.J. and Rusten, K., (2009). Trial translocation of the numbat (*Myrmecobius fasciatus*). *Australian Mammalogy* **31**, 9-16.

Bester, M.N. (1993). *Eradication of cats from sub-Antarctic Marion Island*. Abstracts of the Sixth International Theriological Congress, Sydney, Australia.

Bice, J.K. and Moseby, K.E. (2008). Diet of the re-introduced Greater Bilby (*Macrotis lagotis*; Peramelidae) and Burrowing Bettong (*Bettongia lesueur*; Potoroidae) in the Arid Recovery Reserve, Northern South Australia. *Australian Mammalogy* **30**, 1-12.

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

- Bolam, A.G. (1927). *The Trans-Australian Wonderland*. 6<sup>th</sup> Edition. (University of Western Australia Press: Nedlands).
- Bolles, R.C. (1970). Species-specific defense reactions and avoidance learning. *Psychological Review* **77**, 32-48.
- Bolton, J. and Moseby, K.E. (2004). The activity of Sand Goannas *Varanus gouldii* and their interaction with reintroduced Greater Stick-nest Rats *Leporillus conditor*. *Pacific Conservation Biology* **101**, 193-201.
- Booth, R. and McCracken, H. (1994). *Morbidity and Mortality Report for Eastern Barred Bandicoots *Perameles gunnii* 1990-1994*. Melbourne Zoo, Melbourne.
- Bouskila, A. and Blumstein, D.T. (1992). Rules of thumb for predation hazard assessment: predictions from a dynamic model. *American Naturalist* **139**, 161-176.
- Bowen, Z. and Read, J.L. (1998) Population and demographic patterns of rabbits (*Oryctolagus cuniculus*) at Roxby Downs in arid South Australia and the influence of rabbit haemorrhagic disease. *Wildlife Research* **25**, 655-662.
- Bradbury, J.W. & Vehrencamp, S.L. (1998). *Principles of Animal Communication*. Sunderland, MA, Sinauer.
- Brandle, R., and Moseby, K. E., (1999). Comparative ecology of two populations of *Pseudomys australis* in northern South Australia. *Wildlife Research* **26**, 541-564.
- Bright, P. W. and Morris, P. A. (1994). Animal translocations for conservation: performances of dominance in relation to release methods, origin and season. *Journal of Applied Ecology* **31**, 699-708.
- Burbidge, A.A. (1971). *The fauna and flora of the Monte Bello Islands*. Department of Fisheries and Fauna Western Australia, Report No. 9.
- Burbidge, A.A. (1989). The value of Western Australian islands as biological reservoirs and the development of management priorities. In *Australian and New Zealand islands: nature conservation values and management*, (Ed. A. Burbidge) pp. 17-24. Department of Conservation and Land Management, Perth.
- Burbidge, A.A. (1995). Burrowing Bettong. In *The Mammals of Australia*. (Ed. R. Strahan) pp. 289-291. Reed Books Australia, Chatswood, NSW.
- Burbidge, A.A. and Fuller, P.J. (1979). Mammals of the Warburton region, Western Australia. *Records of the Western Australian Museum* **5**, 1-16.
- Burbridge, A.A. and McKenzie, N.L. (1989). Patterns in the Modern Decline of Western Australia's Vertebrate Fauna: Causes and Conservation Implications. *Biological Conservation* **50**, 143-198.
- Burrows, N.D., Algar, D., Robinson, A.D., Sinagara, J., Ward, B. and Liddelow, G. (2003). Controlling introduced predators in the Gibson Desert of Western Australia. *Journal of Arid Environments* **55**, 691-713.
- Campbell, L. and Croft, D.B. (2001). Comparison of hard and soft release of hand reared eastern grey kangaroos. *Veterinary Conservation Biology*.



- Catling, B. and Reid, A.M. (2003). *Predator and critical weight range species 5. Results of Spring 2002 and Autumn 2003 surveys*. CSIRO Sustainable Ecosystems, Canberra, Australia.
- Catling, P.C. (1988). Similarities and contrasts in the diets of foxes, *Vulpes vulpes*, and cats, *Felis catus*, relative to fluctuating prey populations and drought. *Australian Wildlife Research* 15, 307-17.
- Catling, P.C. and Coman, B.J. (2008). Fox. In: *Mammals of Australia* (Ed. S. Van Dyck and R. Strahan) pp.740-741. Reed New Holland, Sydney, Australia.
- Caughley, G. and Gunn, A. (1996). *Conservation Biology in Theory and Practise*. Blackwell Science, Victoria.
- Chambers, L.K. and Dickman, C.R. (2002). Habitat selection of the long-nosed bandicoot, *Perameles nasuta* (Mammalia, Peramelidae) in a patchy urban environment. *Austral Ecology* 27, 334-342.
- Christensen, P. and Burrows, N. (1994). Project desert dreaming: experimental reintroduction of mammals to the Gibson Desert, Western Australia. In : *Reintroduction Biology of Australian and New Zealand Fauna* (Ed. Serena) pp. 199-207. Surrey Beatty & Sons, Sydney, Australia.
- Clapperton, B.K., Eason, C.T., Weston, R.J., Woolhouse, A.D., and Morgan, D.R. (1994). Development and testing of attractants for feral cats, *Felis catus* L. *Wildlife Research* 21, 389-399.
- Clarke, R.H., Boulton, R.L., Clarke, M.F., (2002). Translocation of the socially complex Black-eared Miner *Manorina melanotis*: a trial using hard and soft release techniques. *Pacific Conservation Biology* 8, 223–234.
- Coman, B.J. and McCutchan, J. (1994). *Predator Exclusion Fencing for Wildlife Management in Australia*. Unpublished report to the Australian Nature Conservation Agency; Canberra.
- Commonwealth of Australia (2007). *Background document for the threat abatement plan for predation by feral cats*. Department of the Environment and Water Resources, Canberra.
- Copley, P. (1988). *The Stick-nest Rats of Australia*. Department of Environment and Planning, South Australia.
- Copley, P. (1991). Feral and domestic cats in South Australia. In *The impact of cats on native wildlife* (Ed. C. Potter) pp 53-59. ANPWS, Canberra.
- Copley, P.B. (1994). Translocations of native vertebrates in South Australia: a review. In: *Reintroduction Biology of Australian and New Zealand Fauna*. (Ed. M. Serena) pp. 35-42.
- Copley, P. (1999). Natural histories of Australia's stick-nest rats, genus *Leporillus* (Rodentia:Muridae). *Wildlife Research* 26, 513-539.
- Corbett, L. (1995). *The Dingo in Australia and Asia*. University of New South Wales Press Ltd, Sydney.
- Corbett, L. K., and Newsome, A.E. (1987). The feeding ecology of the dingo. III. Dietary relationships with widely fluctuating prey populations in arid Australia: an hypothesis of alternation of predation. *Oecologia*, 74, 215–227.

- Coulson, G. and Eldridge, D. (eds) (2010). *Macropods: The Biology of Kangaroos, Wallabies and Rat Kangaroos*. CSIRO Publishing, Collingwood, Victoria.
- Courchamp, F., Langlais, M. and Sugihara, G. (1999). Rabbits killing birds: modelling the hyperpredation process. *Journal of Animal Ecology* **69**, 154-164.
- Cox, J.G. and Lima, S.L. (2006). Naivete´ and an aquatic–terrestrial dichotomy in the effects of introduced predators. *Trends in Ecology and Evolution* **21**, 674–680.
- Crooks, K.R. and Soule, M.E. (1999). Mesopredator release and avifaunal extinctions in a fragmented system. *Nature* **400**, 563-566.
- Curio, E. 1988. Cultural transmission or enemy by birds. In: *Social learning: psychological and biological perspectives*. (Eds. T.R. Zentall and B.G. Galef Jr) pp. 75-97. Lawrence Erlbaum Associates, Hillsdale, New Jersey.
- Daniels, M.J. and Corbett, L. (2003). Redefining introgressed protected mammals: when is a wildcat a wild cat and a dingo a wild dingo? *Wildlife Research* **30**, 213-218.
- Davis, M. H. (1983). Post-release movements of introduced marten. *Journal of Wildlife Management*, 47, 59-66.
- Denniston, C.D. (1978). Small population size and genetic diversity: implications for endangered species. In: *Endangered Birds: Management Techniques for Preserving Threatened Species* (Ed. S. A. Temple) pp. 281-289. Univ. Wisconsin Press, Madison.
- Denny, E.A. and Dickman, C.R. (2010) *Review of Cat Ecology and Management Strategies in Australia*. Invasive Animals Cooperative Research Centre, Canberra.
- Dickman CR (1996). *Overview of the impact of feral cats on Australian native fauna*, report to the Australian Nature Conservation Agency, Canberra.
- Dickman, C.R., Mahon, P.S., Masters, P., and Gibson, D.F., (1999). Long-term dynamics of rodent populations in arid Australia: the influence of rainfall. *Wildlife Research* **26**, 389-403.
- Dodd, C.K. and Seigel, R.A. (1991). Relocation, repatriation and translocation of amphibians and reptiles: are they strategies that work? *Herpetologica* **47**, 336-350.
- Durbin, J. and Watson, G.S. (1971). Testing for Serial Correlation in Least Squares Regression III. *Biometrika* **58**, 1–19.
- Eason, C. (2002). Sodium monofluoroacetate (1080) risk assessment and risk communication. *Toxicology* **181**, 523-230.
- Eason, C.T., Morgan D.R., and Clapperton, B.K. (1992) Toxic bait and baiting strategies for feral cats. In: *Proceedings of the Fifteenth Vertebrate Pest Conference* (Eds. J.E.Borrecco and R.E. Marsh) pp. 371-376. University of California, Davis.
- Edwards J. (2001) *Monitoring vegetation change under different arid management regimes, near Olympic Dam, South Australia, using Landsat TM imagery*. Honours Thesis, Department of Soil and Water, University of Adelaide.

- Edwards, G.P., De Preu, N.D., Shakeshaft, B.J., Crealy, I.V. and Paltridge, R.M. (2001). Home range and movements of male feral cats (*Felis catus*) in a semiarid woodland environment in central Australia. *Austral Ecology* **26**, 93-93.
- Eldridge, S.R., Shakeshaft, B.J. and Nano, T.J. (2002). *The impact of wild dog control on cattle, native and introduced herbivores and introduced predators in central Australia*. Final Scientific Report to Bureau of Rural Sciences, Northern Territory, Australia.
- Engeman, R.M. and Allen, L. (2000). Overview of a passive tracking index for monitoring wild canids and associated species. *International Pest Management Review* **5**, 197-203.
- England, M.H., Ummenhofer, C.C. and Santoso, A., (2006). Interannual rainfall extremes over southwest Western Australia linked to Indian Ocean climate variability. *Journal of Climatology* **19**, 1948e1969.
- Falconer, D.S. (1960). *Introduction to quantitative genetics*. Longman, London.
- Fey, K., Banks, P.B., Ylönen C, H. and Korpimäki, E. (2010). Behavioural responses of voles to simulated risk of predation by a native and an alien mustelid: an odour manipulation experiment. *Wildlife Research* **37**, 273-282.
- Finlayson, G.R., and Moseby, K.E. (2004). Managing confined populations; the influence of density on the home range and habitat use of reintroduced burrowing bettongs (*Bettongia lesueur*). *Wildlife Research* **31**, 457-463.
- Finlayson, G.R., Viera, E.M., Priddel, D., Wheeler, R., Bentley, J., and Dickman, C.R., (2008). Multi-scale patterns of habitat use by re-introduced mammals: A case study using medium-sized marsupials. *Biological Conservation* **141**, 320-331.
- Finlayson, H.H. (1941). On central Australian mammals. Part III. The Muridae. *Transactions of the Royal Society of South Australia* **65**, 215-231.
- Finlayson, H.H. (1961). On central Australian mammals, Part IV. The distribution and status of central Australian species. *Records of the South Australian Museum*, **14**, 141-91.
- Finnis, H.J. (1966). Dr John Harris Browne's Journal of the Sturt Expedition, 1844-5. *South Australiana* **5**, 23-54.
- Fischer, J. and Lindenmayer, D.B., 2000. An assessment of the published results of animal relocations. *Biological Conservation* **96**, 1-11.
- Fisher, P., Algar, R., and Sinagra, J. (1999). Use of Rhodamine B as a systemic bait marker for feral cats (*Felis catus*). *Wildlife Research* **26**, 281 – 285.
- Fleming, P.J.S. (1997). Uptake of baits by red foxes: implications for rabies contingency planning. *Wildlife Research* **24**, 318-346.
- Fleming, P., Corbett, L., Harden, R. and Thomson, P. (2001). *Managing the impacts of dingoes and other wild dogs*. Bureau of Rural Sciences: Canberra, Australia.
- Franklin, I.R. (1980) Evolutionary change in small populations. In: *Conservation Biology* (Eds M.E.Soule and B.A. Wilcox) pp. 135-50. Sinauer, Sunderland, Massachusetts.

- Friedel, M.H. (1985). The population structure and density of central Australian trees and shrubs, and relationships to range condition, rabbit abundance and soil. *Australian Rangeland Journal* **7**, 130-139.
- Friend, J.A. (1990). Status of bandicoots in Western Australia. In: *Bandicoots and Bilbies* (Eds. J. H. Seebeck, P. R. Brown, R. I. Wallis and C. M. Kemper.) pp. 73-84. Surrey Beatty & Sons, Sydney, Australia.
- Friend, J.A. (2008). Numbat (*Myrmecobius fasciatus*). In: *The Mammals of Australia* (Eds. S. Van Dyck, R. Strahan) pp 162-165. Reed New Holland, Australia.
- Friend, J.A., and Algar, D. (1995). *Methods of broad-scale cat control and fox control at a numbat reintroduction site*. Unpublished report to The Australian Nature Conservation Agency, Canberra.
- Gale, J.S. & M.J. Lawrence (1984). The decay of variability. In: *Crop Genetic Resources: Conservation and Evaluation* (Eds. J.H.W. Holden and J.T. Williams) pp. 77–101. Allen and Unwin.
- Galwey, N. W. (2006). *Introduction to Mixed Modelling: Beyond Regression and Analysis of Variance*. J. Wiley and Sons: Chichester, UK.
- Gedeon, C.I., Vászai, O., Koósz, B. and Altbäcker, V. (2011). Morning release into artificial burrows with retention caps facilitates success of European ground squirrel (*Spermophilus citellus*) translocations. *European Journal of Wildlife Research* 10.1007/s10344-011-0504-3.
- Gibson, D.F., Johnson, K.A., Langford, D.G., Cole, J.R., Clarke, D.E. and Willowra Community (1994). The Rufous Hare-wallaby *Lagorchestes hirsutus*: a history of experimental reintroduction in the Tanami Desert, Northern Territory. In: *Reintroduction Biology of Australian and New Zealand Fauna* (Ed. M. Serena.) pp. 171-176. Surrey Beatty & Sons Pty Ltd: Sydney, Australia.
- Gibson, D. F., Lundie-Jenkins, G., Langford, D.G., Cole, J.R. ,Clarke, D.E. and Johnson K.A. (1994). Predation by feral cats, *Felis catus*, on the rufous hare-wallaby, *Lagorchestes hirsutus*, in the Tanami Desert. *Australian Mammalogy* **17**, 103-7.
- Gibson, L.A. (2001). Seasonal changes in diet, food availability and food preference of the greater bilby (*Macrotis lagotis*) in far south-western Queensland. *Wildlife Research* **28**, 121-134.
- Gibson, L. A. and Hume, I. D. (2000). Seasonal field energetics and water influx rates of the greater bilby (*Macrotis lagotis*). *Australian Journal of Zoology* **48**, 225-239.
- Goodrich, J.M. and Buskirk, S.W. (1995). Control of abundant native vertebrates for conservation of endangered species. *Conservation Biology* **9**, 1357-1364.
- Griffin, A.S., Blumstein, D.T., and Evans, C.S. (2000) Training captive-bred or translocated animals to avoid predators. *Conservation Biology* **14**, 1317-1326
- Griffin, A.S., Evans, C.S. and Blumstein, D.T., (2002). Selective Learning in a Marsupial. *Ethology* **108**, 1103-1114

- Griffen, G.F. and Friedel, M.H. (1985). Discontinuous change in central Australia: some implications of major ecological events for land management. *Journal of Arid Environments* **9**, 63-80.
- Griffith, B., Scott, J.M. , Carpenter, J.W. and Reed, C. (1989). Translocation as a species conservation tool: status and strategy. *Science* **245**, 477-480.
- Griffiths, R.A. and Pavajeau, L. (2008). Captive breeding, reintroduction and the conservation of amphibians. *Conservation Biology* **22**, 852-861.
- Grubbs, S.E. and Krausman, P.R. (2009). Observations of coyote-cat interactions. *Journal of Wildlife Management* **73**, 683–685.
- Hairston, N. G., Smith, F. E. and Slobodkin, L. B. (1960), Community structure, population control, and competition. *American Naturalist* **94**, 421–425.
- Hardman, B and Moro, D. (2006) Optimising reintroduction success by delayed dispersal: Is the release protocol important for hare-wallabies? *Biological Conservation* **128**, 403-411.
- Hayward, M.W., Legge, S., Parsons, B., Page, M., Herman, K. And Mulder, E. (2010). Woylie re-introduction as part of the Australian Wildlife Conservancy's endangered species recovery program at Scotia Sanctuary, far western New South Wales, Australia. In: *Global Re-introduction Perspectives. Additional case-studies from around the globe*. (Ed. P.S. Soorae) pp202-207. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi. UAE, xii + 352.
- Hegglin, D., Bontadina, F., Gloor, S., Romer, J., Muller, U., Breitenmoser, U. and Deplazes, P. (2004). Baiting Red Foxes in an Urban Area: A Camera Trap Study. *Journal of Wildlife Management* **68**, 1010-1017.
- Helldin, J.O., Liberg, O. and Gloersen, G. (2006). Lynx (*Lynx lynx*) killing red foxes (*Vulpes vulpes*) in boreal Sweden-frequency and population effects. *Journal of Zoology* **270**, 657-663.
- Henke, S. E. and Bryant, F.C. (1999). Effects of coyote removal on faunal community in western Texas. *Journal of Wildlife Management* **63**, 1066–1081.
- Hirsch, S.M and Bolles, R.C. (1980). On the Ability of Prey to Recognise Predators. *Z. Tierpsychology* **54**, 71-84.
- Hobbs, R.J. (2001). Synergisms among Habitat Fragmentation, Livestock Grazing, and Biotic Invasions in Southwestern Australia. *Conservation Biology* **15**, 1522-1528.
- IUCN, 1998. *Guidelines for re-introductions*. Prepared by IUCN / SSC Reintroduction Specialist Group. IUCN, Gland, Switzerland.
- James, A.I. and Eldridge, D.J. (2007). Reintroduction of fossorial native mammals and potential impacts on ecosystem processes in an Australian desert landscape. *Biological Conservation* doi:10.1016/j.biocon.2007.04.029
- James, A.I., Eldridge, D.J. and Moseby, K.E. (2009), Foraging pits, litter and plant germination in an arid shrubland, *Journal of Arid Environments* doi:10.1016/j.jaridenv.2009.09.016

- Johnson, C. (2006). *Mammal extinction in European Australia*. Cambridge University press, Port Melbourne, Australia.
- Johnson, C.N., Isaac, J.L. and Fisher, D.O. (2007). Rarity of a top predator triggers continent-wide collapse of mammal prey: dingoes and marsupials in Australia. *Proceedings of the Royal Society Biological Sciences* **274**, 341-346.
- Johnson, C. N. and Johnson, K. A. (1983). Behaviour of the Bilby, *Macrotis lagotis* (Reid), (Marsupialia: Thylacomyidae) in captivity. *Australian Wildlife Research* **10**, 77-87.
- Johnson, C. N., and Van Der Wal, J. (2009). Evidence that dingoes limit abundance of a mesopredator in eastern Australian forests. *Journal of Applied Ecology* **46**, 641–646.
- Johnson, K.A. (2008). Bilby. In: *Mammals of Australia*. (Eds. D. Van Dyck and R. Strahan) pp 191-193. Reed New Holland, Sydney, Australia.
- Jones, E. and Coman, B.J. (1982). Ecology of the Feral Cat, *Felis catus* (L.) in South eastern Australia II. Reproduction. *Australian Wildlife Research* **9**, 111-119.
- Jones, F.W. (1924) *The Mammals of South Australia*. South Australian Government Printer, Adelaide.
- Jordan, M.J.R. (2003). Re-introduction protocols and considerations for small mammals. *Re-introduction NEWS* **22**, 39-43.
- Kats, L.B. and Dill, L.M. (1998). The scent of death: chemosensory assessment of predation risk by prey animals. *Ecoscience* **5**, 361-394.
- Kemper, C. (1990). Status of bandicoots in South Australia. In '*Bandicoots and Bilbies*'. (Eds. J. H. Seebeck, P. R. Brown, R. I. Wallis and C. M. Kemper.) pp. 67-72. Surrey Beatty & Sons, Sydney, Australia.
- Kenward, M.G., and Roger, J.H. (1997). Small sample inference for fixed effects from restricted maximum likelihood. *Biometrics* **53**, 983-97.
- King D.R. (1990). *1080 and Australian Fauna*. Western Australian Agricultural Protection Board, Perth.
- King, D.R. and Smith, L.A. (1985). The distribution of the European red fox (*Vulpes vulpes*) in Western Australia. *Records Western Australian Museum* **12**, 197-205.
- Kinnear, J.E., Onus, M.L. and Bromilow, R.N. (1988). Fox Control and Rock-Wallaby Population Dynamics. *Australian Wildlife Research* **15**, 435-450.
- Kinnear, J.E., Onus, M.L., and Sumner, N.R. (1998). Fox control and rock-wallaby population dynamics - II. An update. *Wildlife Research* **25**, 81-88.
- Kinnear, J.E., Sumner, N.R. and Onus, M.L. (2002). The red fox in Australia—an exotic predator turned biocontrol agent. *Biological Conservation* **108**, 335-359.
- Kitchener, D.J & E. Vicker (1981). *Catalogue of Modern Mammals in the Western Australian Museum 1895 to 1981*. Page(s) 184. WA Museum: Perth.
- Kleiman, D.G. (1989). Reintroduction of captive mammals for conservation. Guidelines for introducing endangered species into the wild. *Bioscience* **39**, 152-161.

- Lande, R. and Barrowclough, G.F. (1987). Effective population size, genetic variation and their use in population management. In: *Viable Populations for Conservation* (Ed. M.E. Soule) pp. 87-123. Cambridge University Press, Cambridge.
- Lange, R.T. and Graham, C.R. (1983). Rabbits and the failure of regeneration in Australian arid zone *Acacia*. *Australian Journal of Ecology* **8**, 377-381.
- Letnic, M. (2007). The impacts of pastoralism on the fauna of arid Australia. In *Animals of arid Australia: out on their own?* (Eds C. R. Dickman, D. Lunney and S. Burgin) pp. 65–75. Sydney, Australia, Royal Zoological Society of New South Wales.
- Letnic, M., Crowther, M.S. and Koch, F. (2009). Does a top-order predator provide an endangered rodent with refuge from an invasive mesopredator? *Animal Conservation* **2009**, 1-11.
- Letnic, M., Koch, F., Gordon, C., Crowther, M.S. and Dickman, C.R. (2009). Keystone effects of an alien top-predator stem extinctions of native mammals. *Proceedings of the Royal Society- Biological Sciences* **276**, 3249-3256.
- Letty, J., Marchandean, S., Clobert, J. and Aubineau, J. (2000). Improving translocation success; an experimental study of anti-stress treatment and release method for wild rabbits. *Animal Conservation* **3**, 211–219.
- Lewis, J.C., Sallee, K.L. and Golightly, J.R.T. (1999). Introduction and Range Expansion of Nonnative Red Foxes. *The American Midland Naturalist* **142**, 372-377.
- Lindenmayer, D.B. and Likens, G.E. (2009). Adaptive monitoring: a new paradigm for long-term research and monitoring. *Trends in Ecology and Evolution* **24**, 482-6.
- Lindenmayer, D.B., Gibbons, P., Bourke, M., Burgman, M., Dickman, C.R., Ferrier, S., Fitzsimons, J., Freudenberger, D., Garnett, S.T., Groves, C., Hobbs, R.J., Kingsford, R.T., Krebs, C., Legge, S., Lowe, A.J., McLean, R., Montambault, J., Possingham, H., Radford, J., Robinson, D., Smallbone, L., Thomas, D., Varcoe, T., Vardon, M., Wardle, G., Woinarski, J. and Zenger, A. (2012). Improving biodiversity monitoring. *Austral Ecology* **37**, 285-294.
- Lindstrom, E. R., Brainerd, S.M. Helldin, J.O. and Overskaug, K. (1995). Pine marten–red fox interactions: a case of intraguild predation? *Annales Zoologici Fennici* **32**, 123–130.
- Long, K. and Robley, A. (2004). *Cost Effective Feral Animal Exclusion Fencing for Areas of High Conservation Value in Australia*. Australian Government Report, Australia.
- Lunney, D. (2001). Causes of the extinction of native mammals of the Western Division of New South Wales: an ecological interpretation of the nineteenth century historical record. *Rangeland Journal* **23**, 44-70.
- Lunney, D., Trigg, B. Eby, P. and Ashby, E. (1990). Analysis of scats of dogs, *Canis familiaris*, and foxes, *Vulpes vulpes* (Canidae Carnivora), in coastal Bega, New South Wales. *Australian Wildlife Research* **17**, 61-68.
- Lyne, A.G. and CSIRO Division of Wildlife and Rangelands Research (1982). The bandicoots *Isoodon macrourus* and *Perameles nasuta*: their maintenance and breeding in captivity. In: *The Management of Australian Mammals in Captivity*. (Ed. D. D. Evans.) pp. 47-52. Zoological Board of Victoria, Melbourne, Australia.

- MacCracken, J.G. (1982). Coyote food in a Southern California suburb. *Wildlife Society Bulletin* **10**, 280–281.
- MacDonald, D.W. (1977). On food preference in the red fox. *Mammal Review* **7**, 7-23.
- Magurran, A.E. (1989). Acquired Recognition of Predator Odor in the European Minnow (*Phoxinus phoxinus*) *Ethology* **82**, 216-223.
- Mahon, P.S., Banks, P.B., and Dickman, C.R. (1998). Population indices for wild carnivores: a critical study in sand-dune habitat, south-western Queensland. *Wildlife Research* **25**, 11–22.
- Marks, C.A., and Bloomfield, T.E. (1999). Bait uptake by foxes (*Vulpes vulpes*) in urban Melbourne: the potential of oral vaccination for rabies control. *Wildlife Research* **26**, 777-787.
- Marlow, N.J., Thomson, P.C., Algar, D. Rose, K. Kok, N.E. and Sinagra, J.A. (2000). Demographic characteristics and social organisation of a population of red foxes in a rangeland area in Western Australia. *Wildlife Research* **27**, 457-64.
- Marsack, P., & Campbell, G. (1990). Feeding behavior and diet of dingoes in the Nullabor region, Western Australia. *Wildlife Research* **17**, 349-57.
- Martin, G.R., Twig, L.E., and Robinson, D.J. (1996). Comparison of the diet of feral cats from rural and pastoral Western Australia. *Wildlife Research* **34**, 475-84.
- Masters, P., (1993). The effects of fire-driven succession and rainfall on small mammals in spinifex grassland at Uluru National Park, Northern Territory. *Wildlife Research* **20**, 803-813.
- Matějů, J., Štěpánka Říčanová, Simona Poláková, Michal Ambros, Borys Kala, Kristýna Matějů and Lukáš Kratochvíl (2011). Method of releasing and number of animals are determinants for the success of European ground squirrel (*Spermophilus citellus*) reintroductions. *European Journal of Wildlife Research* **58**, 473-482.
- Matson, T.K., Goldizen, A.W. and Jamran, P.J. (2004). Factors affecting the success of translocations of the black-faced impala in Namibia. *Biological Conservation* **116**, 359-365
- McCallum, H., Timmers, P. and Hoyle, S. (1995). Modelling the Impact of Predation on Reintroductions of Bridled Nailtail Wallabies. *Wildlife Research* **22**, 163-171.
- McCracken (1983). *Reproduction in the female bilby, Macrotis lagotis*. BSc. (Vet.) (Honours) Thesis, University of Sydney, New South Wales.
- McCracken (1990). *Reproduction in the Greater Bilby, Macrotis lagotis (Reid)-a comparison with other perameloids*. In: Bandicoots and Bilbies (Ed. J.H. Seebeck) pp 199-204.
- McIlroy, J.C. (1984). The sensitivity of Australian animals to 1080 poison VII. Native and introduced birds. *Australian Wildlife Research* **11**, 373-85.
- McIlroy, J.C. (1985). The sensitivity of Australian animals to 1080 poison VIII. Amphibians and reptiles. *Australian Wildlife Research* **12**, 113-18.
- McIlroy, J.C. and King, D.R. (1990). Appropriate amounts of 1080 poison in baits to control foxes, *Vulpes vulpes*. *Australian Wildlife Research* **17**, 11-13.
- McLean, I.G., Holzer, C. and Studholme, B.J.S. (1999). Teaching predator-recognition to a native bird. Implications for management. *Biological Conservation* **87**, 123-130.



- McLean, I.G., Lundie-Jenkins, G. and Jarman, P.J. (1996). Teaching an endangered mammal to recognise predators. *Biological Conservation* **75**, 51-62.
- McLean, I.G., Schmitt, N.T., Jarman, P.J., Duncan, C., and Wynne, C.D.L. (2000) Learning for life: Training marsupials to recognize introduced predators. *Behaviour* **137**, 1361-1376.
- Michell, B.D. and Banks, P.B. (2005). Do wild dogs exclude foxes? Evidence for competition from dietary and spatial overlaps. *Austral Ecology* **30**, 581-591.
- Miller, B., Biggins, D., Wemmer, C., Powell, R., Calvo, L., Hanebury, L and Wharton, T., (1990). Development of Survival Skills in Captive- Raised Siberian Polecats (*Mustela eversmanni*) 11: Predator Avoidance. *Journal of Ethology* **8**, 95-104
- Mineka, S. and Cook, M. (1988). Social learning and the acquisition of snake fear in monkeys. In: *Social Learning: Psychological and Biological Perspectives*. L. Erlbaum, Hillsdale, New Jersey, pp, 51-73. *Cited in:* Griffin, A.S., Evans, C.S., Blumstein, D.T., (2002). Selective Learning in a Marsupial. *Ethology* **108**, 1103-1114.
- Mitchell, N., Haeffner, R., Veer, V., Fulfold-Gardner, M., Clerveaux, W., Veitch, C.R., and Mitchell, G. (2002). Cat eradication and restoration of endangered iguanas (*Cyclura carinata*) on Long Cay, Caicos Bank, Turks and Caicos Islands, British West Indies. In: *Turning the Tide: The eradication of invasive species. IUCN SSC Invasive Species Specialist Group* (Eds. C. R. Veitch and M. N. Clout.) pp. 206-212. IUCN, Gland, Switzerland and Cambridge, UK.
- Molsher, R.L., Dickman, C., Newsome, A.E. and Muller, W. (2005). Home ranges of feral cats (*Felis catus*) in central-western New South Wales, Australia. *Wildlife Research* **32**, 587-595.
- Molsher, R., Newsome, A. and Dickman, C. (1999). Feeding ecology and population dynamics of the feral cat (*Felis catus*) in relation to the availability of prey in central-eastern New South Wales. *Wildlife Research* **26**, 593-607.
- Moro, D. (2001). Update of translocation of Thevenard Island mice to Serrurier Island, Western Australia: success and excess. *Reintroduction News* **20**, 17-18.
- Moro, D. (2003). Translocation of captive-bred dibblers *Parantechinus apicalis* (Marsupialia: Dasyuridae) to Escape Island, Western Australia. *Biological Conservation* **111**, 305-315.
- Morris, K., (2000). Fauna translocations in Western Australia 1971-1999: an overview. In: *Biodiversity and the Reintroduction of Native Fauna at Uluru-Kata Tjuta National Park* (Eds. J.S. Gillen, R. Hamilton, W.A. Low, C. Creagh) pp 64-74. The Bureau of Rural Sciences, Kingston ACT.
- Morton, S.R. (1990). The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia* **16**, 201-213.
- Moseby K.E. and Bice, J. (2004). A trial reintroduction of the Greater Stick-nest Rat (*Leporillus conditor*) in arid South Australia. *Ecological Management and Restoration* **5**, 118-124.

- Moseby, K.E., Hill, B.M. & Read, J.L. (2009). Arid Recovery- A comparison of reptile and small mammal populations inside and outside a large rabbit, cat and fox-proof enclosure in arid South Australia. *Austral Ecology* **34**,156-169
- Moseby, K.E., O'Donnell, E. (2003). Reintroduction of the greater bilby, *Macrotis lagotis* (Reid) (Marsupialia: Thylacomyidae), to northern South Australia: survival, ecology and notes on reintroduction protocols. *Wildlife Research* **30**, 15-27.
- Moseby, K.E., Owens, H., Brandle, R, Bice, J.K., Gates, J., (2006). Variation in population dynamics and movement patterns between two geographically isolated populations of the dusky hopping mouse (*Notomys fuscus*). *Wildlife Research* **33**, 222-232.
- Moseby, K.E. and Read, J.L. (2006). The efficacy of feral cat, fox and rabbit exclusion fence designs for threatened species protection. *Biological Conservation* **127**, 429-437.
- Moseby, K.E., Read, J.L. and Gee, P. (1998). *A study of the Davenport Range black-footed rock wallaby colony and possible threatening processes*. Final Report to the Wildlife Conservation Fund, Department for Environment and Heritage, Adelaide, Australia.
- Moseby, K.E., Stott, J. and Crisp, H. (2009b). Improving the effectiveness of poison baiting for the feral cat and European fox in northern South Australia: The influence of movement, habitat use and activity. *Wildlife Research* **36**, 1-14.
- Neumann, D. R. 1999. Agonistic behavior in harbor seals (*Phoca vitulina*) in relation to the availability of haul-out space. *Marine Mammal Science* **15**, 507-525.
- New, T.R (2000) *Conservation Biology: An introduction to Southern Australia*. Oxford University press pp 312-316.
- Newsome A. E. (1971). The ecology of red kangaroos. *Australian Journal of Zoology* **16**, 32–50.
- Newsome, A. (1990). The control of vertebrate pests by vertebrate predators. *Trends in Ecology and Evolution* **5**, 187-91.
- Newsome, A.E. (2001). The biology and ecology of the dingo, In: *A Symposium on the Dingo* (Eds. C.R. Dickman & D. Lunney) pp. 20-33. Royal Zoological Society of New South Wales, Sydney.
- Newsome, A. E., Catling, P. C., Cooke, B. D. and Smyth, R. (2001). Two ecological universes separated by the dingo fence in semi-arid Australia: interactions between landscapes, herbivory and carnivory, with and without dingoes. *Rangeland Journal* **23**, 71–98.
- Newsome, A. E., and Coman B.J. (1989). Canids. In: *Fauna of Australia. Mammalia Vol. 1B* (Eds D.W. Walron and B.J. Richardson) pp 993-1005. Australian Government Publishing Service, Canberra, Australia.
- Newsome, A.E., and Corbett, L.K. (1975). Outbreaks of rodents in semi-arid and arid Australia: causes, preventions, and evolutionary considerations. In: *Rodents and Desert Environments* (Eds. I. Prakash and P.K. Ghosh) pp117-153. The Hague, Netherlands.
- Newsome, A.E., Parer, I. and Catling, P.C. (1989). Prolonged prey suppression by carnivores - predator-removal experiments. *Oecologia* **78**, 458-67.

- Noble, J.C., Muller, W.J., Detling, J.K. and Pfitzner, G.H. (2007). Landscape ecology of the burrowing bettong: Warren distribution and patch dynamics in semiarid eastern Australia. *Austral Ecology* **32**, 326-337.
- Nolte, D.L., Mason, J.R., Epple, G., Aronov, E. and Campbell, D.L. (1994). Why are predator urines aversive to prey? *Journal of Chemical Ecology* **20**, 1505-1516.
- Norrdahl, K., and Korpimäki, E. (1999). Does mobility or sex of voles affect risk of predation by mammalian predators? *Ecology* **79**, 226-232.
- Olsson, M., Wapstra, E., Swan, G., Snaith, E., Clarke, R. and Madsen, T. (2005). Effects of long-term fox baiting on species composition and abundance in an Australian lizard community. *Austral Ecology* **30**, 899-905.
- Owens, H.M. and Read, J.L. (1999). *Mammals of the Lake Eyre South region*. Vol. 1, Part 2 in: (W.J.H. Slater ed.) Lake Eyre South Monograph Series, RGSSA, Adelaide.
- Palmer R (1996a). *Feral Pest Program Project 12, Cat research and management in Queensland, year 3; and Project 43. Dispersal and spatial organisation of cats on the Diamantina Plains, year 2*. Unpublished report to the Feral Pests Program, Australian Nature Conservation Agency, Canberra.
- Palmer R (1996b). *Feral Pest Program Project 12, Cat research and management in Queensland, year 4; and Project 43. Dispersal and spatial organisation of cats on the Diamantina Plains, year 3*. Unpublished report to the Feral Pests Program, Australian Nature Conservation Agency, Canberra, Interim Final Report, May 1996.
- Paltridge, R.M. (2002). The diet of cats, foxes and dingoes in relation to prey availability in the Tanami Desert, Northern Territory. *Wildlife Research* **29**, 389-403.
- Paltridge, R.M., Gibson, D., and Edwards, G. (1997). Diet of the feral cat (*Felis catus*) in Central Australia. *Wildlife Research* **24**, 67-76.
- Parmenter, R. R., and MacMahon, J. A. (1983) Factors determining the abundance and distribution of rodents in a shrub-steppe ecosystems: The role of shrubs. *Oecologia* **59**, 145-56.
- Pedler, L. and Copley, P.B. (1993). *Reintroduction of Stick-nest Rats to Reevesby Island, South Australia*. Biological Conservation Branch, South Australian Dept of Environment and Land Management: Adelaide.
- Perot-Sinal, T.S., Ossenkopp, K.P. and Kavaliers, M. (1999). Brief predator odour exposure activates the HPA axis independent of locomotor changes. *Neurotheology* **10**, 775-780.
- Peterson, R. O. (1995). Wolves as interspecific competitors in canid ecology. In: *Ecology and conservation of wolves in a changing world*. (Eds. L.N. Carbyn, S.H. Fritts and D.R. Seip) pp 315-323. Canadian Circumpolar Institute, Alberta, Canada.
- Pettigrew, J. D. (1993). A burst of feral cats in the Diamantina – a lesson for the management of pest species? In: *'Cat management workshop proceedings'* (Eds. G. Siepen and C. Owens). Queensland Department of Environment and Heritage, Brisbane.

- Pople, A.R., Grigg, G.C. Cairns, S.C. Beard, L.A. and Alexander, P. (2000). Trends in the numbers of red kangaroos and emus on either side of the South Australian dingo fence: Evidence for predator regulation? *Wildlife Research* **27**, 269-76.
- Priddel, D. and Wheeler, R. (1997). Efficacy of fox control in reducing the mortality of released captive-reared Malleefowl, *Leipoa ocellata*. *Wildlife Research* **24**, 469-482.
- Priddel, D. and Wheeler, R. (2002). *An experimental translocation of Brush-tailed Bettongs Bettongia penicillata to western New South Wales*. Department of Conservation and Environment, Hurstville, NSW, Australia.
- Priddel, D., and Wheeler, R. (2004). An experimental translocation of brush-tailed bettongs (*Bettongia penicillata*) to western New South Wales. *Wildlife Research* **31**, 421-432.
- Quinn, T. (1997). Coyote (*Canis latrans*) food habits in three urban habitat types of western Washington. *Northwest Science* **71**, 1-5.
- R Development Core Team (2012). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL <http://www.R-project.org/>.
- Read, J.L. (1994). A Retrospective View of the Quality of the Fauna Component of the Olympic Dam Project Environmental Impact Statement. *Journal of Environmental Management* **41**, 167-185.
- Read, J.L. (1995). Recruitment characteristics of the White Cypress Pine (*Callitris glaucophylla*) in arid South Australia. *The Rangeland Journal* **17**, 228-240.
- Read, J.L., (2004). Catastrophic drought-induced die-off of perennial chenopod shrubs in arid Australia following intensive cattle browsing. *Journal of Arid Environments* **58**, 535-544.
- Read, J.L. (2010). Can fastidiousness kill the cat? The potential for target-specific poisoning of feral cats through oral grooming. *Ecological Management and Restoration* **11**, 230-233.
- Read, J.L. and Bowen, Z. (2001). Population dynamics, diet and aspects of the biology of feral cats and foxes in arid South Australia. *Wildlife Research* **28**, 195-203.
- Read, J.L., Carter, J., Moseby, K.M. and Greenville, A. (2008). Ecological roles of rabbit, bettong and bilby warrens in arid Australia. *Journal of Arid Environments* **72**, 2124-2130.
- Read, J.L., and Cunningham, R. (2010). Relative impacts of cattle grazing and feral animals on an Australian arid zone reptile and small mammal assemblage. *Austral Ecology* **35**, 314-324.
- Read, J.L., Ebdon F.R., and Donohoe, P. (2000). The terrestrial birds of the Roxby Downs Area: a ten year history. *South Australian Ornithologist* **33**, 71-83.
- Read, J.L., and Eldridge, S. (2010). An optimised rapid detection technique for simultaneously monitoring activity of rabbits, cats, foxes and dingoes in the rangelands. *Rangeland Journal* **32**, 389-394.
- Read, J.R., Johnston, G.R. and Morley, T.P. (2011). Snake predation thwarts trial reintroduction of Woma Pythons. *Oryx* **45**, 505-512.

- Read, J.L., Kovak, K.J., Brook, B.W. and Fordham, D.A. (2012). Booming during a bust; Asynchronous population responses of arid zone lizards to climatic variables. *Acta Oecologica* **40**, 51-61.
- Read, J.L., Moseby, K.E., Briffa, J., Kilpatrick, A.D. and Freeman, A. (2011). Eradication of rabbits from landscape scale exclosures: pipedream or possibility? *Ecological Management and Restoration* **12**, 46-53.
- Read, J.L. and Ward, M.J. (2011). Bringing back warru: initiation and implementation of the South Australian Warru Recovery Plan. *Australian Mammalogy* **33**, 214–220.
- Richards, J.D. and Short, J. (2003). Reintroduction and establishment of the western barred bandicoot *Perameles bougainville* (Marsupialia: Peramelidae) at Shark Bay, Western Australia. *Biological Conservation* **109**, 181-195.
- Risbey, D.A., Calver, M.C. and Short, J. (1997). Control of Feral cats for nature conservaton. I. Field tests of four baiting methods. *Wildlife Research* **24**, 319-326.
- Risbey, D., Calver, M.C., and Short, J. (1999). The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. I. Exploring potential impact using diet analysis. *Wildlife Research* **26**, 621–630.
- Robley, A.J., Short, J. and Bradley, S. (2001). Dietary overlap between the burrowing bettong (*Bettongia lesueur*) and the European rabbit (*Oryctolagus cuniculus*) in semi-arid coastal Western Australia. *Wildlife Research* **28**, 341-349.
- Robley, A.J., Short, J. and Bradley, S. (2002). Do European rabbits (*Oryctolagus cuniculus*) influence the population ecology of the Burrowing Bettong (*Bettongia lesueur*)? *Wildlife Research* **29**, 423-429.
- Rosatte, R., Hamr, J., Ranta, B., Young, J. & Cool, N. (2002). *Elk restoration in Ontario, Canada: infectious disease management strategy, 1998-2001*. Annals of the New York Academy of Sciences, Domestic Animal/Wildlife Interface: Issue for Disease Control, Conservation, Sustainable Food Production, and Emerging Diseases, 969, 358-365.
- Ryan, S., Moseby, K.E. and Paton, D.C. (2003). Comparative foraging preferences of the Greater Stick Nest Rat *Leporillus conditor* and the European Rabbit *Oryctolagus cuniculus*: implications for regeneration of arid lands. *Australian Mammalogy* **25**, 135-146.
- Saji, N.H., Ambrizzi, T. and Ferraz, S.E.T. (2005). Indian Ocean Dipole mode events and austral surface air temperature anomalies. *Dynamic Atmospheric Oceans* **39**, 87-101.
- Saji, N.H., Goswami, B.N., Vinayachandran, P.N., Yamagata, T. (1999). A dipole mode in the tropical Indian Ocean. *Nature* **401**, 360-362.
- Santer, U., Short, J. and Turner, B. (1997). Social organisation and warren use of the Burrowing Bettong, *Bettongia lesueur* (Macropodoidea:Potoroidae). *Wildlife Research* **24**, 143-157.
- Sarrazin, F. and Barbault, R. (1996). Reintroduction: challenges and lessons for basic ecology. *Trends in Ecology and Evolution* **11**, 474-478.

- Saunders, G., Coman, B., Kinnear, J., and Braysher, M. (1995). *Managing Vertebrate Pests: Foxes*. Australian Government Publishing Service, Canberra.
- Scott, L.K. Hume, I.D. and Dickman, C.R. (1999). Ecology and population biology of long-nosed bandicoots (*Perameles nasuta*) at North Head, Sydney Harbour National Park. *Wildlife Research* **26**, 805-821.
- Seddon, P. J. (1999). Persistence without intervention: assessing success in wildlife reintroductions. *Trends in Ecology and Evolution* **14**, 503.
- Seddon, P. J., Armstrong, D. P. and Maloney R. F. (2007). Developing the science of reintroduction biology. *Conservation Biology* **21**, 303-312.
- Seddon P.J., Soorae P.S. and Launay, F. (2005). Taxonomic bias in reintroduction projects. *Animal Conservation* **8**, 51-58.
- Seebeck, J.H. (1979). Status of the Barred Bandicoot *Perameles gunnii*, in Victoria: with a Note on Husbandry of a Captive Colony. *Australian Wildlife Research* **6**, 255-264.
- Seebeck, J.H. (1990). Recovery Management of the Eastern Barred Bandicoot in Victoria: Statewide Strategy. In: *Management and Conservation of Small Populations: Conference Proceedings in Melbourne, Australia, September 26-27, 1989*. (Eds. T. W. Clark and J. H. Seebeck.) pp. 165-177. Chicago Zoological Society (Ill.), Brookfield, USA.
- Seebeck, J. and Booth, R. (1996). Eastern Barred Bandicoot recovery: the role of the veterinarian in the management of endangered species. *Australian Veterinary Journal* **73**, 81-83.
- Shaffer, M.L. (1990). Population viability analysis. *Conservation Biology* **4**, 39–40.
- Shargo, E. S. (1988). *Home range, movement, and activity patterns of coyotes (Canis latrans) in Los Angeles suburbs*. Dissertation, University of California, Los Angeles, USA.
- Sharp, A., Copley, P., Bignall, J., Carthew, S., Taggart, D., Van Weenen, J., Johnson, G., Smith, I., Swales, J., Kemp, L., Austin, T. and Rudd, K. (2010). Re-introduction of the “extinct in the wild” South Australian mainland Tammar Wallaby on Yorke Peninsula, Australia. In: *Global reintroduction perspectives: Additional case studies from around the globe*. (Ed. Soorae, P. ) pp 208-214. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi, UAE.
- Sherwin, W.B. and Brown, P.R. (1990). Problems in the estimation of the effective population size of the Eastern Barred Bandicoot *Perameles gunnii* at Hamilton, Victoria. In: *Bandicoots and Bilbies*. (Eds. J. H. Seebeck, P. R. Brown, R. L. Wallis and C. M. Kemper.) pp. 367-374. Surrey Beatty & Sons, Sydney, Australia.
- Short, J. (1998). The extinction of rat-kangaroos (Marsupialia: Potoroidae) in New South Wales, Australia. *Biological Conservation* **86**, 365-377.
- Short, J. (2009). *The characteristics and success of vertebrate translocations within Australia*. Australian Government Department of Agriculture, Fisheries and Forestry, Canberra.

- Short, J., Bradshaw, S.D., Giles, J., Prince, R.I.T. and Wilson, G.R. (1992). Reintroduction of macropods (Marsupialia: Macropodoidea) in Australia A review. *Biological Conservation* **62**, 189-204.
- Short, J., Kinnear, J.E. and Robley, A. (2002). Surplus killing by introduced predators in Australia—evidence for ineffective anti-predator adaptations in native prey species? *Biological Conservation* **103**, 283-301.
- Short J., Richards J.D. and Turner B. (1998). Ecology of the Western Barred Bandicoot (*Perameles bougainville*) (marsupialia: Peramelidae) on Dorre and Bernier Islands, Western Australia. *Wildlife Research* **25**, 567-86.
- Short, J. and Smith, A. (1994). Mammal decline and recovery in Australia. *Journal of Mammalogy* **75**, 288-297.
- Short, J. and Turner, B. (1999). Ecology of burrowing bettongs, *Bettongia lesueur* (Marsupialia: Potoroidae), on Dorre and Bernier Islands, Western Australia. *Wildlife Research* **26**, 651-669.
- Short, J. and Turner, B. (2000). Reintroduction of the burrowing bettong *Bettongia lesueur* (Marsupialia: Potoroidae) to mainland Australia. *Biological Conservation* **96**, 185-196.
- Short, J., Turner, B., Risbey, D. A. and Carnamah, R. (1997). Control of feral cats for nature conservation II. Population reduction by poisoning. *Wildlife Research* **24**, 703–714.
- Short, J., Turner, B., and Risbey, D.A. (2002). Control of feral cats for Nature Conservation. III. Trapping. *Wildlife Research* **29**, 475-487.
- Sinclair, A. R. E., Pech, R. P., Dickman, C. R., Hik, D., Mahon, P. and Newsome, A. E.(1998). Predicting effects of predators on conservation of endangered prey. *Conservation Biology* **12**, 1313-17.
- Smith, A.P. and Quin, D.G. (1996). Patterns and causes of extinction and decline in Australian conilurine rodents. *Biological Conservation* **77**, 243-267.
- Snyder, N.F.R., Derrickson, S.R., Beissinger, S.R., Wiley, J.W., Smith, T.B., Toone, W.D. and Miller, B.J. (1996). Limitations of captive breeding in endangered species recovery. *Conservation Biology* **10**, 338-348.
- Soderquist, T.R. (1994). The importance of hypothesis testing in reintroduction biology: examples from the reintroduction of the carnivorous marsupial *Phascogale tapoatafa*. In: *Reintroduction Biology of Australian and New Zealand Fauna*. (Ed. M. Serena) pp. 159-164. Surrey Beatty & Sons, Chipping Norton.
- Soorae, P.S. (2010). *Global Re-introduction Perspectives: 2010. Additional case-studies from around the globe*. IUCN/SSC Re-introduction Specialist Group, Abu Dhabi, UAE.
- Southgate, R. (1994). Why reintroduce the Bilby? In: *Reintroduction Biology of Australian and New Zealand Fauna*. (Ed. M. Serena.) pp. 165-170. Surrey Beatty & Sons: Chipping Norton.
- Southgate, R., Bellchambers, K., Romanow, K. and Whitfield, S. (1994). *Reintroduction of the Greater Bilby A Field Guide*. Final Report to World Wide Fund for Nature (Australia), Northern Territory, Australia.

- Southgate R.I., Christie P. and Bellchambers K. (2000). Breeding biology of captive, reintroduced and wild greater bilbies, *Macrotis lagotis* (Marsupialia: Peramilidae). *Wildlife Research* **27**, 621-628.
- Southgate, R., McRae, P. and Atherton, R. (1995). Trapping techniques and a pen design for the Greater Bilby, *Macrotis lagotis*. *Australian Mammalogy* **18**, 101-104.
- Southgate, R. and Possingham, H. (1995). Modelling the reintroduction of the greater bilby *Macrotis lagotis* using the metapopulation model. Analysis of the likelihood of extinction. *Biological Conservation* **73**, 151-160.
- Spencer, P.B.S. (1991). Evidence of predation by a feral cat, *Felis catus* (Carnivora: Felidae) on an isolated rock-wallaby colony in tropical Queensland. *Australian Mammalogy* **14**, 143-144.
- Steen, H. (1994). Low survival of long distance dispersers of the root vole (*Microtus oeconomus*). *Annales Zoologici Fennici* **31**, 271-274.
- Sugiura, N. (1978). Further analysis of the data by Akaike's information criterion and the finite corrections. *Communications in Statistics, Theory and Methods* **A7**, 13-26
- Sullivan, T.P. and Crump, D.R. (1984). Influence of mustelid scent gland compounds on suppression of feeding by snowshoe hares (*Lepus americanus*) *Journal of Chemical Ecology* **10**, 1809-1821.
- Sutherland, W. J. (ed.) (1998). *Conservation Science and Action*. Blackwell Science, Oxford.
- Sutherland, W.J., Armstrong, D., Butchart, S.H.M., Earnhardt, J.M., Ewen, J. Jamieson, I., Jones, C.G., Lee, R., Newbery, P., Nichols, J.D., Parker, K.A., Sarrazin, F., Seddon, P.J., Shah, N. and Tatayah, V. (2010). Standards for documenting and monitoring bird reintroduction projects. *Conservation Letters* **3**, 229-235.
- Tammie K. Matsona, Anne W. Goldizena, Peter J. JarmanMgurrana, A.E. (1989) Acquired Recognition of Predator Odor in the European Minnow (*Phoxinus phoxinus*) *Ethology* **82**, 216-223.
- Taraborelli, P.A., Dacar, M., and Gianonni, S.N. (2003). Effect of plant cover on seed removal by rodents in the Monte Desert (Mendoza, Argentina). *Austral Ecology* **28**, 651-657.
- Taylor, R.H. (1979). How the Macquarie Island parakeet became extinct. *New Zealand Journal of Ecology* **2**, 42-5.
- Taylor, R.J. (1992). Seasonal changes in the diet of the Tasmanian Bettong (*Bettongia gaimardi*) a mycophagous marsupial. *Journal of Mammalogy* **73**, 408-414.
- Teixeira, C. P., C. S. de Azevedo, M. Mendl, C. F. Cipreste, and Young, R.J. (2007). Revisiting translocation and reintroduction programmes: the importance of considering stress. *Animal Behaviour* **73**, 1-13.
- Thompson, J.R., Bleich, V.C., Torres, S.G. and Mulcahy, G.P. (2001). Translocation techniques for mountain sheep: does the method matter? *The Southwestern Naturalist* **46**, 87-93.



- Thomson, P.C. (1992a) The behavioural ecology of dingoes in north-western Australia. III. hunting and feeding behaviour and diet. *Wildlife Research* **19**, 531-41.
- Thomson, P.C. (1992b). The behavioural ecology of dingoes in north-western Australia. IV. social and spatial organisation, and movements. *Wildlife Research* **19**, 543-63.
- Thomson, P.C. and Algar, D. (2000). The uptake of dried meat baits by foxes and investigations of baiting rates in Western Australia. *Wildlife Research* **27**, 451-456.
- Thomson, P.C., Marlow, N.J., Rose, K., and Kok, N.E. (2000). The effectiveness of a large-scale baiting campaign and an evaluation of a buffer zone strategy for fox control. *Wildlife Research* **27**, 465-472.
- Twyford, K.L., Humphrey, P.G., Nunn, R.P. and Willoughby, L. (2000). Eradication of Feral Cats (*Felis catus*) from Gabo Island, south-east Victoria. *Ecological Management and Restoration* **1**, 42-49.
- Tyndale-Biscoe, C.H. (1968). Reproduction and post-natal development in the marsupial *Bettongia lesueur* (Quoy & Gaimard). *Australian Journal of Zoology* **16**, 577-602.
- Vale, T., Schmitz, A., Fogarty, P. and Bentley, J. (2004). *Translocation Proposal for the reintroduction of the Burrowing Bettong or Boodie (Bettongia lesueur) to Yookamurra Sanctuary Murraylands South Australia*. Report, Australia.
- Van Polanen Petel, M. A., Marks, C.A., and Morgan, D.G. (2001). Bait palatability influences the caching behaviour of the red fox (*Vulpes vulpes*). *Wildlife Research* **28**, 395-401.
- Veitch, C.R. (1985). Methods of eradicating feral cats from offshore islands in New Zealand. In: *Conservation of island birds* (Ed. P.J. Moors) pp 125-141. International Council for Bird Preservation, Cambridge.
- Wallach, A.D., Johnson, C.N., Ritchie, E.G. and O'Neill, A.J. (2010). , Predator control promotes invasive dominated ecological states. *Ecology Letters* **13**, 1008-1013.
- Wallach, A.D., Ritchie, E.G., Read, J and O'Neill, A.J. (2009). More than mere numbers: the impact of lethal control on the social stability of a top-order predator. *PLoS ONE*, **4**, e6861. doi:10.1371/journal.pone.0006861
- Wanless, R.M., Cunningham, J., Hockey, P.A.R., Wanless, J., White, R.W. and Wiseman, R. (2002). The success of a soft-release reintroduction of the flightless Aldabra rail (*Dryolimnas [cuvieri] aldabranus*) on Aldabra Atoll, Seychelles. *Biological Conservation* **107**, 203–210.
- Warren, K., Swan, R., Bodetti, T., Friend, T., Hill, S. and Timms, P. (2005). Ocular *Chlamydiales* infections of Western barred Bandicoots (*Perameles bougainville*) in Western Australia. *Journal of Zoo and Wildlife Medicine* **36**, 100-102.
- Watson, A.J. and Halley, M. (1999). *Recovery Plan for the Eastern Barred Bandicoot Perameles gunnii (mainland subspecies)*. Department of Natural Resources and Environment, Victoria
- White, G. C., and Garrott, R. A. (1990). Results of radio tracking eastern barred bandicoots, *perameles gunnii*, (marsupialia: Peramelidae) at Gellibrand Hill Park Victoria. *Australian Mammalogy* **16**, 51-54.

- Wilson, A.D. (1990). The effect of grazing on Australian ecosystems. In: *Australian Ecosystems: 200 years of utilization, degradation and reconstruction*. (Eds. D.A. Saunders, A.J.M. Hopkins, R.A.How) pp 235-244. Proceedings of the Ecological Society of Australia, Surrey Beatty and Sons, Chipping Norton, Australia.
- Wimberger, K., Downs, C.T. and Perrin, M.R. (2009). Two unsuccessful re-introductions of rock hyraxes (*Procavia capensis*) into a reserve in the KwaZulu-Natal Province, South Africa. *South African Journal of Wildlife Research* **39**, 192-201.
- Wolf, C.M., Garland, T Jr. and Griffith, B. (1998). Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. *Biological Conservation* **86**, 243-255.
- Wolf, C.M., Griffith, B., Reed, C. and Stanley, A. (1996). Avian and Mammalian Translocations: Update and Reanalysis of 1987. *Conservation Biology* **10**, 1142-1154.
- Wood, B., Tershy, B.R., Hermosillo, M.A., Donlan, C.J., Sanchez, J.A., Keitt, B.S., Croll, D.A., Howald, G.R., and Biavaschi, N. (2002). Removing cats from islands in north-west Mexico. In: *Turning the Tide: The eradication of invasive species*. (Eds. C. R. Veitch and M. N. Clout.) pp 374-380. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK.
- Wood, S.N. (2004). Stable and efficient multiple smoothing parameter estimation for generalized additive models. *Journal of the American Statistical Association* **99**, 673-686.
- Wood, S.N. (2006). *Generalized Additive Models: an introduction with R*, CRC.
- Wood, S.N. (2011). Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. *Journal of the Royal Statistical Society (B)* **73**, 3-36.
- Ylönen, H. (2001). Predator odours and behavioural responses of small rodents: an evolutionary perspective. In: *Advances in Vertebrate Pest Management II*. (Eds. H.J. Pelz, P. D. Cowan and C. J. Feare.) pp. 123–138. Filander, Fürth, Germany.