Fauna assemblages of the spinifex sand plains in central Australia:
response to climate, fire and predation

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# CONTENTS

List of Tables ........................................................................................................ iv
List of Figures ........................................................................................................ vi
List of Appendices ................................................................................................. viii
Thesis Declaration ................................................................................................. ix
Acknowledgements ............................................................................................... x

Abstract .................................................................................................................. xii

**CHAPTER 1 - Introduction** ................................................................................. 1

1.1. Arid ecosystems .............................................................................................. 3
    1.1.1. Australia’s arid zone ............................................................................. 4
1.2. Additional pressures on fauna decline ......................................................... 6
    1.2.1. Fire ....................................................................................................... 6
    1.2.2. Land use/Agriculture ........................................................................ 7
    1.2.3. Introduced feral predators ................................................................. 8
    1.2.4. Introduced feral herbivores ............................................................... 8
1.3. Persistence of Australian arid fauna ............................................................ 9
    1.3.1. Brush-tailed mulgara (Dasycercus blythi) ....................................... 11
1.4. Managing and supporting persistence of Australia’s arid fauna .............. 13
1.5. Reducing the impacts of additional pressures ............................................. 13
    1.5.1. Monitoring species to assess the effectiveness of management ...... 15
1.6. Project aims .................................................................................................. 16

**CHAPTER 2 - Study sites, species assemblages and recent history** .......... 19

2.1. Study sites ..................................................................................................... 20
    2.1.1. Location .............................................................................................. 20
    2.1.2. Climate .............................................................................................. 21
    2.1.3. Vegetation assemblages ................................................................. 25
2.2. Vertebrate assemblages .............................................................................. 27
    2.2.1. Mammals .......................................................................................... 27
    2.2.2. Reptiles ............................................................................................ 29
2.3. Recent history of species assemblages and land management .......... 32
    2.3.1. Management at Newhaven Wildlife Sanctuary ......................... 34
    2.3.2. Management at the Borefields ..................................................... 35

**CHAPTER 3 - Species occurrence in a post boom period; Part A:**
**Small mammals and reptiles** ......................................................................... 37

3.1. Introduction .................................................................................................. 39
3.2. Methods ....................................................................................................... 42
    3.2.1. Study Site ......................................................................................... 42
    3.2.2. Small mammal and reptile sampling ......................................... 43
    3.2.3. Invertebrate sampling ................................................................. 44
CHAPTER 5 - Refuge use in the brush-tailed mulgara (*Dasycercus blythi*) ......................................................... 103
  5.1. Introduction .................................................................................................................... 105
  5.2. Methods .......................................................................................................................... 108
    5.2.1. Study sites and rainfall data ...................................................................................... 108
    5.2.2. *Dasycercus blythi* sampling .................................................................................. 109
    5.2.3. Statistical analysis .................................................................................................. 113
  5.3. Results ............................................................................................................................ 117
  5.4. Discussion ....................................................................................................................... 122
    5.4.1. Management considerations ................................................................................. 125
CHAPTER 6 Habitat use of the brush-tailed mulgara (*Dasycercus blythi*) ................................................................. 127

6.1. Introduction ........................................................................................................................................................................ 129
6.2. Methods .................................................................................................................................................................................. 132
   6.2.1. Study site and species ...................................................................................................................................................... 132
   6.2.2. Trapping protocol ............................................................................................................................................................ 133
   6.2.3. Habitat use and availability .......................................................................................................................................... 134
   6.2.4. Statistical analysis ......................................................................................................................................................... 138
6.3. Results ..................................................................................................................................................................................... 140
   6.3.1. Habitat structure ............................................................................................................................................................ 141
   6.3.2. Movement pathway ...................................................................................................................................................... 146
6.4. Discussion .............................................................................................................................................................................. 148

CHAPTER 7 - Efficacy of three popular techniques for detecting and monitoring arid zone species in a spinifex sand plains habitat ...................................................... 155

7.1. Introduction ............................................................................................................................................................................ 157
7.2. Methods .................................................................................................................................................................................. 160
   7.2.1. Study sites .................................................................................................................................................................... 160
   7.2.2. Live trapping ................................................................................................................................................................. 162
   7.2.3. Sign surveys ................................................................................................................................................................. 162
   7.2.4. Infra-red (IR) camera surveys .................................................................................................................................. 163
   7.2.5. Statistical analysis ....................................................................................................................................................... 164
7.3. Results ..................................................................................................................................................................................... 169
   7.3.1. Capture rates by methods ............................................................................................................................................. 169
   7.3.2. Species richness between methods .............................................................................................................................. 171
   7.3.3. Probability of detection between methods .................................................................................................................. 174
   7.3.4. Cost analysis ............................................................................................................................................................... 175
7.4. Discussion .............................................................................................................................................................................. 176
   7.4.1. Effect of varying deployment methods ....................................................................................................................... 177
   7.4.2. Survey methods ............................................................................................................................................................ 180
   7.4.3. Cost analysis ............................................................................................................................................................... 182
   7.4.4. Management recommendations .................................................................................................................................. 182

CHAPTER 8 - General discussion ................................................................................................................................. 185

8.1. Overview .............................................................................................................................................................................. 186
8.2. Persistence in a variable climate ........................................................................................................................................ 187
8.3. Identifying species specific requirements for *Dasycercus blythi* .......................................................... 189
8.4. Improving monitoring methodology .............................................................................................................................. 191
8.5. A pathway to better management ................................................................................................................................. 192
8.6. Conclusion ........................................................................................................................................................................... 195

Appendices ........................................................................................................................................................................... 197

References .............................................................................................................................................................................. 223
LIST OF TABLES

Table 2.1  Mammal species recorded or potentially occurring within the spinifex sand plain habitat at Newhaven Wildlife Sanctuary and the Borefields ........................................................................................................ 27

Table 2.2  Reptile species recorded or potentially occurring within the spinifex sand plain habitat Newhaven Wildlife Sanctuary and the Borefields ............................................................ 29

Table 3.1  Climate variables used in generalised additive mixed models for each response variable examined ........................................................................................................................................................................ 48

Table 3.2  Grouping of variables for candidate model structure based upon ecological relationship to response variables. ........................................................................................................ 49

Table 3.3  Mammal and reptile captures during all surveys between November 2012 and November 2014 ........................................................................................................................................................................ 51

Table 3.4  Spearman’s rank-order correlations between vertebrate ordinal abundance (# individuals) across all surveys........................................................................................................................................................................ 52

Table 3.5  Model selection summary statistics from small mammal GAMMS with considerable support ........................................................................................................................................................................ 56

Table 3.6  Model selection summary statistics of reptile GAMMS with considerable support ........................................................................................................................................................................ 62

Table 4.1  Climate variables used in generalised additive mixed models for each response variable examined ........................................................................................................................................................................ 83

Table 4.2  Grouping of variables for candidate model structure based upon ecological relationship to response variables. ........................................................................................................................................................................ 83

Table 4.3  Invertebrate captures during all surveys between November 2012 and November 2014 ........................................................................................................................................................................ 86

Table 4.4  Spearman’s rank-order correlations between invertebrate ordinal abundance (# individuals) across all surveys........................................................................................................................................................................ 87

Table 4.5  Model selection summary statistics of invertebrate GAMMS with considerable support ........................................................................................................................................................................ 90

Table 5.1  Summary of trapping and sign surveys conducted at Newhaven Wildlife Sanctuary between 2008 and 2015 ........................................................................................................................................................................ 109

Table 5.2  Trapping regime used for D. blythi surveys at Uluru Kata–Tjuta National Park between 2009 and 2011 ........................................................................................................................................................................ 112
Table 5.3 Rainfall time periods utilised in relation to when \textit{D. blythi} were sampled and explanation of the related ecological influence. ........................................115

Table 5.4 Covariate structures of candidate models tested for \textit{D. blythi} occurrence..........................................................................................................................116

Table 5.5 Generalised linear mixed effect model outcomes for all candidate models tested..........................................................................................................................................................................................................................121

Table 5.6 Generalised linear mixed model (GLMM) estimates for the top model for predicting \textit{D. blythi} occurrence..................................................................................................................121

Table 6.1 Description of habitat variables surveyed along spool lines..........137

Table 6.2 Sex, body mass, reproductive condition and fire category of successful spooling events..........................................................................................................................................................................................................................141

Table 6.3 Habitat and \textit{D. blythi} characteristics of areas of high use determined through spool and line tracking.................................................................148

Table 7.1 Number of reptile and mammal ‘captures’ for each survey and deployment technique across eight survey periods during 2014. .................................169

Table 7.2 Model outcomes and detection probabilities (±SE) for selected species across sampling methods..........................................................................................................................................................................................................................174
LIST OF FIGURES

Figure 1.1 Map of global distribution of climatic zones ................................................. 3

Figure 1.2 Spatial distribution of arid and semi-arid areas across Australia................. 4

Figure 1.3 Current (< 50 years old) and historical (> 50 years old) occurrence records of mulgara (Dasycercus blythi) within Australia................................. 12

Figure 2.1 Newhaven Wildlife Sanctuary, Ayers Rock Resort and Uluru Kata-Tjuta National Park boundary showing major roads and towns. ............... 21

Figure 2.2 Monthly rainfall (mm) and temperature variables (°C) at Newhaven Wildlife Sanctuary prior and during study period............................... 23

Figure 2.3 Monthly rainfall (mm) and temperature variables (°C) at the Borefields prior to and during study period............................................... 24

Figure 2.4 Vegetation classification within Newhaven Wildlife Sanctuary and UKJNP and Ayres Rock Resort.............................................................. 25

Figure 3.1 Site locations across Newhaven Wildlife Sanctuary............................ 43

Figure 3.2 Percentage of sites across Newhaven Wildlife Sanctuary with predator sign detected over all survey periods........................................... 53

Figure 3.3 Total monthly rainfall and average minimum and maximum monthly temperatures at Newhaven Wildlife Sanctuary over and prior to all survey periods................................................ 54

Figure 3.4 Change over all survey periods of species richness of Dasyuridae and Rodentia and total captures of P. desertor and P. hermannsburgensis across all sites........................................ 55

Figure 3.5 Smoothing plots for non-climate variables present in top GAMMS for small mammal........................................................................ 58

Figure 3.6 Smoothing plots for climate variables present in top GAMMS for small mammals............................................................................... 59

Figure 3.7 Change over all survey periods for species richness of Agamidae, Elapidae, Geckonidae, Pygopidae, Scincidae, Thyllopidae and Varanidae and abundance of C. hanloni, C. schomburgkii, L. bipes, L. labialis and N. ornatus across all surveys........................................ 60

Figure 3.8 Smoothing plots for non-climate variables present in top GAMMS for reptiles............................................................. 64

Figure 3.9 Smoothing plots for climate variables present in top GAMMS for reptiles............................................................. 65

Figure 4.1 Mean small predator abundance (±SD) per site across all survey periods............................................................. 89
Figure 4.2 Smoothing plots for non-climate variables present in top GAMMS for invertebrates

Figure 4.3 Smoothing plots for climate variables present in top GAMMS for invertebrates.

Figure 5.1 Location of live trapping and sign survey sites at Newhaven Wildlife Sanctuary between 2008 and 2015.

Figure 5.2 Proportion of sites *D. blythi* detected at varying rainfall levels the previous season.

Figure 5.3 Frequency of *D. blythi* detection at sites located within prime habitat during periods of low rainfall at UKTNP.

Figure 5.4 Frequency of *D. blythi* detection at sites located within prime habitat during periods of low rainfall at Newhaven Wildlife Sanctuary.

Figure 6.1 (a) Newhaven Wildlife Sanctuary boundary showing location of successful (closed circles) and unsuccessful (open circles) *D. blythi* trap sites.

Figure 6.2 Spool positioning on *D. blythi* rumph.

Figure 6.3 Average Manly-Chesson selection indices (±SD) for all (a) vegetation and (b) substrate types.

Figure 6.4 Duality diagram of eigen analysis of vegetation selection ratios of *D. blythi* at Newhaven Wildlife Sanctuary.

Figure 6.5. Duality diagram of eigen analysis of substrate selection ratios of *D. blythi* at Newhaven Wildlife Sanctuary.

Figure 6.6 Average quadrat cover score (QCM) and maximum and minimum distance to vegetation (DTV) for used and available locations including and excluding shelter sites.

Figure 6.7 Comparison of change in bearing across vegetation and substrate types for all spooling events.

Figure 7.1 Site locations for all surveys conducted at Newhaven Wildlife Sanctuary and Yulara borefields.

Figure 7.2 Site layout of live traps, IR camera traps and sign survey boundaries at Newhaven Wildlife Sanctuary.

Figure 7.3 Capture rates of mammal and reptiles for each deployment method used within live trapping, camera traps and sign surveys.

Figure 7.4 Comparison of Survey species richness (SSR) and Accumulative species richness (ASR) for Total/All species, Target species Target mammal species and Target reptile species.

Figure 7.5 Accumulated total survey costs of three comparative survey methods over 12 survey periods.
# LIST OF APPENDICES

**Appendix 1** List of all candidate models for mammal and reptile diversity and individual abundance. ................................................................. 198

**Appendix 2** All mammal candidate model outcomes. ................................................. 199

**Appendix 3** All reptile candidate model outcomes. .................................................... 201

**Appendix 4** List of all candidate models for invertebrate ordinal diversity and individual abundance. ........................................................................ 204

**Appendix 5** All invertebrate candidate model outcomes. ............................................. 205

**Appendix 6** Site layout of live traps during annual surveys at Newhaven Wildlife Sanctuary. .................................................................................. 210

**Appendix 7** Location of live trapping and sign survey sites at Uluru Kata – Tjuta National Park between 2000 and 2011. .................................................. 211

**Appendix 8** Additional methods tested to assess microhabitat selection by *D. blythi* in the spinifex sand plains at Newhaven Wildlife Sanctuary. .............. 215

**Appendix 9** Maps showing the four pathways of spool lines identified as areas of intense use ...................................................................................... 217

**Appendix 10** Total captures of mammal species/groups by survey method and deployment technique ................................................................. 218

**Appendix 11** Total captures of reptile species/group by survey method and deployment technique ................................................................. 219

**Appendix 12** Detailed costings for all survey methods assessed ................................. 221
THESIS DECLARATION

I hereby declare that the work herein, now submitted as a thesis for the degree of Doctor of Philosophy at Charles Darwin University, is the result of my own investigations, and contains no material previously published or written by another person, except where due reference has been made in the text. I hereby certify that the work contains no material that has been accepted for the award of any other degree of diploma in any university or other tertiary institution.

I give consent to this copy of my thesis, when deposited in the University Library, being made available for loan and photocopying online via the University’s Open Access repository eSpace.

Signature    Date 05/06/2017
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ABSTRACT

Australia’s arid zone is one of the most unique desert regions in the world due to its dominance across the continent, distinctive rainfall patterns, low soil nutrients and unique faunal assemblage. Drivers of faunal assemblages are varied, with rainfall, fire and predators considered most influential. The aim of this study was to identify the drivers of fauna persistence following a period of high rainfall and to refine management strategies in the spinifex sand plains of central Australia, with particular reference to the brush-tailed mulgara, *Dasycercus blythi*. I utilised broad scale fauna monitoring to assess the effect of environmental factors on persistence generally and conducted a detailed assessment of the fine-scale habitat use by *D. blythi*. I further compared the efficacy of three popular monitoring techniques utilised in the region.

This study revealed large mammalian predators were most influential in predicting the occurrence of small mammals and reptile assemblages, whilst rainfall and temperature were most influential for invertebrates. The distribution of *D. blythi* populations contracted spatially during periods of low rainfall to areas of potential refuge. Fine scale habitat use indicating access to food resources was more influential than selecting areas with low exposure to predators. Infrared cameras sampled the broadest range of species and were the most cost effective over time, whilst sign surveys were most cost effective short term and live trapping essential in detecting several specific species.

Overall, these findings inform land managers on the importance of gaining a better understanding of the invertebrate fauna in the region and implementing effective predator control during periods of low rainfall. By targeting management to specific areas of refuge land managers could reduce financial
costs whilst maintaining support for key species, such as *D. blythi*. Land managers would further benefit from implementing a greater variety of monitoring techniques to maximise the effectiveness of monitoring both financially and statistically.
CHAPTER 1

INTRODUCTION

Photo: Finding the work life balance (A Molyneux)
Preface
This thesis improves our understanding of the factors influencing the occurrence and detection of Australia’s arid zone fauna in the spinifex sand plains habitat. One of the primary drivers of ecosystem functioning in Australia’s arid zone is the ‘boom and bust’ cycle which describes the fluctuation from high to low rainfall (Morton et al. 2011). These fluctuations in rainfall are mirrored in the abundance and/or distribution of fauna with many species experiencing dramatic declines as rainfall reduces (Dickman et al. 2010; Pavey & Nano 2013b; Read et al. 2012). The extent of these declines are further influenced by several biotic and abiotic factors including fire, agriculture and the presence of introduced competitors and predators (McKenzie et al. 2007).

My thesis examines the direct effect of fire, predation and climate on the occurrence of fauna during the early stages of bust in the spinifex sand plains of central Australia. It investigates how a key species in the region, the brush-tailed mulgara (*Dasycercus blythi*), utilises its surroundings to persist during a period when pressures are at their highest and assesses the efficacy of several monitoring techniques in the region.

In the first chapter, I give an overview of arid systems and describe the unique conditions experienced in Australia’s arid zone. I discuss the effects exacerbating species declines in a post boom period and give insight into how *D. blythi* has persisted whilst many other similar species have gone extinct in the region. I describe and discuss current management strategies and review the techniques utilised to assess changes in species abundance in the region. Lastly, I present the aims of my thesis, which is followed by the second chapter; a detailed description of the study sites used throughout the project.
1.1. Arid ecosystems

Arid ecosystems are considered extreme in their climate with high summer maximum temperatures (40 - 45°C) and low winter temperatures (0 °C and below; Perry & Goodall 1979). Although arid and semi-arid regions cover approximately 37% of the world’s land surface, only 13% of the world’s population live in them (Figure 1.1; Warner 2004).

Most arid systems are considered water-controlled due to extremely low annual rainfall (0- 250mm) and high temperatures that increase evapotranspiration and salinity (Noy-Meir 1973; Schwinning et al. 2004). Since the mid-20th century, our understanding of the effect of rainfall in water-controlled environments has developed considerably. Initially the pulse-reserve concept by Noy-Meir (1973) explained the increased productivity periodically seen in arid systems was a response to pulses in rainfall and its cascading effect through trophic pathways. This concept has expanded over time to include the existence of water partitioning in the soil bank (Ryel et al. 2004), response thresholds (Beatley 1974) and response time lags (Schwinning et al. 2004). In addition, we have also gained a greater understanding on the critical influence of nutrient...
availability, particularly nitrogen, on ecosystem functioning in many arid and semi-arid regions (Morton et al. 2011; Whitford & Wade 2002). This understanding of what drives arid ecosystems has aided our comprehension not only regarding the communal drivers of arid systems globally but also the variations found in separate geographical areas.

1.1.1. **Australia’s arid zone**
Covering 70% of its land mass, Australia has the largest proportion of arid and semi-arid habitat of any continent in the world (Figure 1.2; James et al. 1999). These regions support a variety of different habitats including sandy plains/dunes, stony (gibber) plains, mountain ranges, riverine corridors and floodplains. The most widespread habitat in the country is sandy plains/dunes, which covers nearly 50% of Australia’s arid zone. These areas have very high drainage and little ability to maintain large bodies of water, making them one of the driest desert habitats (Oldfield 2004; Orians & Milewski 2007).

![Figure 1.2 Spatial distribution of arid (orange) and semi-arid (yellow) areas across Australia (Trewin 2008)](image)

Figure 1.2 Spatial distribution of arid (orange) and semi-arid (yellow) areas across Australia (Trewin 2008)
The primary ecosystem drivers in Australia’s arid zone are thought to be unpredictable rainfall and soil composition (Morton et al. 2011), particularly in the northern half of the continent (Van Etten 2009). The soil in Australian arid regions is considered nutrient poor and rainfall events are particularly unpredictable, with large differences in rainfall found over months, years and decades (Schwinning et al. 2004). Large rainfall events produce pulses in primary productivity and a dynamic response by arid zone fauna. These pulses (colloquially known as booms) are well documented as they cascade through trophic pathways resulting in a dramatic change across all levels of ecosystem dynamics and functioning (e.g. Dickman et al. 2010; Letnic & Dickman 2006; Letnic et al. 2005; Pavey & Nano 2013b; Read et al. 2012).

Responses to rainfall events vary considerably between individual animal groups. Large increases in rodent abundances are commonly recorded (e.g. Dickman et al. 2010; Greenville et al. 2012; Pavey & Nano 2013a; Predavec 1994) whilst carnivorous marsupials (dasyurids) display somewhat less dramatic increases in the abundance (dasyurids; Dickman et al. 2001; Gibson & Cole 1992). The lack of a large response in many dasyurid species is thought to be due to their fixed life history that inhibits their ability to reproduce quickly (Dickman et al. 2001). Although reptiles are known to respond positively to rainfall (Read et al. 2012) factors such as vegetation structure (Letnic et al. 2004) and invertebrate abundance (Pastore et al. 2013) are also considered key determinants for assemblage and abundance. The least well-studied fauna group is invertebrates that are represented in only a handful of studies from Australia’s arid zone (e.g. Kwok & Eldridge 2015; Langlands et al. 2006; Palmer 2010). Overall climate is considered an important driver of many orders (Barrow et al. 2007; Letnic et al. 2004; Palmer 2010); however, site specific variations in
vegetation (Kwok & Eldridge 2015; Langlands et al. 2006; Smith & Morton 1990) have also been found to be influential.

As rainfall reduces and resources decline, there is often a rapid post boom decline of many species which results in a subset of species persisting during periods of low resource availability (Letnic & Dickman 2006; Letnic & Dickman 2005; Letnic et al. 2004). Highly mobile species often track available resources whilst others remain in isolated patches, which are capable of maintaining relatively high resource availability (Haythornthwaite & Dickman 2006b; Pavey & Nano 2013a). Following these declines, individuals that remain become harder to detect reducing our ability to gain accurate information and therefore make informed management decisions. Furthermore, declines are often exacerbated by additional pressures, which become more pronounced, as rainfall declines.

1.2. Additional pressures on fauna decline

Since European settlement Australia’s arid zone has experienced one of the most dramatic declines and extinction rates of biodiversity in recent history (Chisholm & Taylor 2010). The drivers of these declines are fire, agriculture, introduced predators and competitors, and the rapid anthropomorphic changes that have occurred (McKenzie et al. 2007). These changes, along with natural climate driven fluctuations, have created unique dynamic patterns of species persistence in the landscape that vary across both time and space.

1.2.1. Fire

Fire has been an important component of Australia’s ecological history, including lightning induced wildfires (Burrows et al. 2006; Burrows & van Didden 1991) as well as the human-induced fires by indigenous people (Latz &
Green 1995). Predominantly considered ‘cool’ fires, the latter tended to be small and patchy, providing serial stages of vegetation recovery in the landscape as well as fire breaks for larger fires (Bowman 1998; Yibarbuk et al. 2001). Since European settlement indigenous fire regimes have decreased or disappeared resulting in a greater occurrence of large scale intense fires, particularly after rainfall, due to greater biomass and connectivity of fuel loads (Bird et al. 2012). These fires have considerable effects on many aspects of faunal assemblages including direct impacts such as death of individuals or destruction of important resources such as burrows or nest sites (Chew et al. 1959; Letnic et al. 2004). Fires also cause many indirect impacts including altering the availability of food, reducing habitat continuity, increasing exposure to predators as well as attracting more predators (Letnic & Dickman 2005; McGregor et al. 2014; Schultz 2006; Woodrow 1992). The effect of these impacts is further complicated through species-specific responses that can vary with fire intensity, frequency and extent (Chalmandrier et al. 2013; Griffiths et al. 2015; Lawes et al. 2015; Pastro et al. 2014).

1.2.2. Land use/Agriculture
Agricultural lands cover approximately 52% of Australia’s land mass with a large portion of Australia’s arid zone being utilised for agriculture in the form of large cattle stations that average over 3000 km² (Centralian Land Management Association 2011). Grazing by cattle, *Bos taurus*, has been implicated in the decline of many species in central Australia (Dickman et al. 1993; Eldridge & Kwok 2008; Gibson & Cole 1996; Haythornthwaite & Dickman 2006a; James et al. 1999; Morton et al. 2011). The trampling and grazing by cattle can increase erosion and reduce vegetation cover, ultimately decreasing the availability of suitable habitat for native species (Letnic & Dickman 2010). The development
of agriculture has also interacted with other pressures, such as the increased
permanent water sources used in agriculture facilitating the survival of
introduced predators across a broader distribution (Letnic & Dickman 2006).

1.2.3. Introduced feral predators
Hyper predation during post boom periods, particularly from introduced
predators, is one of the primary driving factors in population declines during
this period (Meserve et al. 2003). Across Australia two introduced predators
have been strongly identified as influential in the decline of small and medium-
sized mammals in the arid zone; the feral house cat, *Felis catus*, and European
red fox, *Vulpes vulpes* (Dickman 1996a; Dickman et al. 1993; Letnic et al. 2005;
Mahon 1999).

Cats and foxes occur across Australia’s entire arid zone (Dickman 1996a). Feral
cats are known to be versatile and opportunistic in their diets. However, the
European rabbit, *Oryctolagus cuniculus*, and small mammals usually dominate
their diet when available (Paltridge 2002; Pavey et al. 2008). Foxes are known to
prey readily on rabbits in most arid areas yet are also considered to be
opportunistic and feed upon a variety of native vertebrate and invertebrate
species (Cupples et al. 2011; Jarman 1986). The influence of predators is further
confounded by their interaction with several other pressures, as described
above.

1.2.4. Introduced feral herbivores
Australia’s arid zone currently supports several introduced herbivores, including
horses (*Equus caballus*), rabbits, donkeys (*Equus africanus asinus*), goats
(*Capra hircus*) and camels (*Camelus dromedarius*). Although many are limited
in their distribution due to a reliance on water (e.g. James et al. 1999;
Landsberg et al. 1997) and/or specific habitats (i.e. rocky areas and goats; Letnic et al. 2015), many still occur in stable populations across the arid zone. The subterranean lifestyle and rapid reproduction of the European rabbit means it is capable of surviving in high abundances in even the driest areas (Williams et al. 1995). Rabbits not only directly compete with native herbivores for food but also increase erosion and can reduce the survival and regeneration of many plants species through their extensive burrows (Leigh et al. 1987).

Like rabbits, camels are capable of persisting across the entire arid zone. Australian population now considered the largest wild population of dromedary camels in the world (Saalfeld & Edwards 2010). Considering their high abundance, the erosion caused by camels is minimal due to their soft-padded feet, except in areas of high use such as water sources (Edwards et al. 2010). However, due to their size, varied diet and abundance, overgrazing and competition are the biggest threats camels impose on native species (Brim-Box et al. 2010; Edwards et al. 2010).

1.3. Persistence of Australian arid fauna

In comparison to other continents, Australia’s arid zone fauna includes relatively few large native herbivores and a high diversity of reptiles and small mammals, particularly carnivorous marsupials (dasyurids; Byrne et al. 2008).

Reptiles, particularly the family Scincidae (skinks), are considered the most successful in exploiting the desert environment due to their ectothermic physiology (Laity 2008). In addition, a large diversity of mammal, bird, amphibian and invertebrate life has also been recorded (e.g. Byrne et al. 2008; Kwok et al. 2016; Landsberg et al. 1999; Letnic & Dickman 2006; Letnic et al. 2011; Read et al. 2012). Across all these groups, a variety of both behavioural
and physiological adaptations has evolved to withstand the arid zones’ harsh conditions. Some of the most common adaptations of desert mammals (and some reptile species, such as *Liopholis kintorei*) are burrowing and/or nocturnal activity patterns which reduce exposure to extreme daytime temperatures (Laity 2008). More mobile animals, such as birds or large mammals, are capable of moving significant distances in order to exploit resource rich patches and avoid extreme seasonal variation (Oldfield 2004; Ruiz-Esparza et al. 2011). Whilst most reptiles are considered diurnal due to the ectothermic physiology, a variety of adaptations to the extreme conditions has also evolved. Several species, particularly in the family *Scincidae* (skinks) display bi-modal activity patterns (active early morning and late afternoon) or preferences for more closed habitats, that would provide shelter from extreme conditions (Pianka 1969a, 1969b). More mobile animals, such as birds or large mammals, are capable of moving considerable distances in order to exploit resource rich patches and avoid extreme seasonal variation (Ehrhardt et al. 2005; Geiser & Masters 1994; Geiser & Pavey 2007; Körtner & Geiser 2011; Pavey & Geiser 2008).

Physiological adaptations are also common. Many smaller species utilise daily or seasonal torpor which reduce the amount of resources required to survive (Ehrhardt et al. 2005; Geiser & Masters 1994; Geiser & Pavey 2007; Körtner & Geiser 2011; Pavey & Geiser 2008). Other species show more specialised adaptations such as the ability to store fat or to excrete highly concentrated urine to retain water (Geiser 2004; Macmillen & Lee 1969).

The amount of information available on the life history of specific species in Australia’s arid zone is varied. Unlike many temperate zone species which have been heavily studied, arid zone species are often cryptic and display extremely low capture rates (< 5%; Bos & Carthew 2001; Dickman et al. 2001; Morton et
al. 1989; Moseby & Read 2001). Small sample sizes and low, scattered populations have restricted conclusions on many individual species to periods of high rainfall when detection rates are highest. Improving our understanding on the life history of species that continue to persist in the region (such as the brush-tailed mulgara) is essential for improving ongoing conservation management in the region.

1.3.1. **Brush-tailed mulgara (Dasycercus blythi)**
One of the few small to medium-sized Australian mammals that has persisted across its distribution despite increasing pressures is the brush-tailed mulgara (*Dasycercus blythi*). Scientific confidence in the status of the *D. blythi* has recently seen its conservation status downgraded from Vulnerable to Least Concern (Woinarski et al. 2014). *D. blythi* is one of the largest remaining native marsupial predators (50 – 120 grams) found in the sand plains of western and central Australia (Van Dyck et al. 2008). It is strictly nocturnal and while known to inhabit a variety of habitats (e.g. Pavey et al. 2011), is found predominantly in spinifex sand plains habitat (Woolley 1990). It digs its own burrows as well as uses the burrows of a variety of other species (Dickman 1996c; Masters & Dickman 2012; Woolley 1990). *D. blythi* uses the same area for several years and is considered a generalist feeder; consuming a variety of insects, arthropods and small vertebrates (Chen et al. 1998; Masters 1998). Invertebrates are the predominant prey choice with specific selection between invertebrate orders varying between populations and seasons (Chen et al. 1998; Masters 1998). During periods of high resource availability *D. blythi* store fat in their tails and can fluctuate dramatically in body weight and condition depending on resource availability and breeding requirements (Masters 1998). Individuals can enter daily torpor to reduce energetic requirements which
further aids in maintaining stable home ranges when resources are scarce (Chen et al. 1998; Körtner et al. 2008).

Previous studies have indicated *D. blythi* are generally solitary; however, male and female pairs have been known to share burrows during the breeding season (e.g. Masters 2003). Little is known about the breeding strategies of *D. blythi* in the wild. Available records indicate a monoestrous breeding cycle that occurs between May and September producing up to six young per litter (Gibson & Cole 1992). Young suckle until they become independent at 3 - 4 months and reach reproductive maturity at 10 - 11 months (Woolley 2005). Captive records show breeding can continue to up to 5 years indicating the species is fairly long lived in the dasyuridae family (Woolley 1973).

Recent reclassification in *Dasycercus* group into 2 distinct species, *D. blythi* and *D. cristicauda* (Woolley 2005), has created ambiguity in regards their historical distribution. Currently the *D. blythi* is thought to occur in stable, scattered populations (Figure 1.3; Gibson & Cole 1992; Masters 1993; Masters 1998; Masters & Dickman 2012). The density and distribution of *D. blythi* populations are largely driven by longer-term climatic conditions (El Niño-Southern Oscillation; ENSO). It is known to experience smaller scale annual declines during the breeding season followed by an increase late in the year caused by the emergence of juveniles into the population (Masters & Dickman 2012). The main threats to *D. blythi* are currently the presence of feral species, particularly predators such as cats and foxes and increased exposure to these predators brought about through changes in fire regimes (Burbidge & McKenzie 1989; Dickman et al. 1993; Greenville et al. 2016; Körtner et al. 2007; Morton 1990). Recently, it has been proposed that its successful persistence, compared to many other similar species in the region, is a result of a varied diet, ability to enter
torpor and ability to withstand large fluctuations in body weight (Masters & Dickman 2012).

Figure 1.3 Current (< 50 years old; black) and historical (> 50 years old; grey) occurrence records of mulgara (*Dasycercus blythi*) in Australia (Atlas of Living Australia 2016)

1.4. **Managing and supporting persistence of Australia’s arid fauna**

Since European settlement many Australian arid zone species have experienced decline or extinction (Woinarski et al. 2007). One of the most dramatic declines has been in the small to medium sized mammals (35g – 5500g) in the southern arid zone and the Western Australian wheat belt (Woinarski et al. 2014; Woinarski et al. 2007). In the southern arid zone, central Australia currently has 29 species classified as extinct or threatened (Woinarski et al. 2007). As over half these species are classified in the orders, Dasyuromorphia and Rodentia, they are an important focus for conservation management in the region.

1.5. **Reducing the impacts of additional pressures**
Currently, conservation management strategies primarily focus on reducing the effects of factors contributing to species declines. Fire is being increasingly utilised by land managers across Australia’s arid zone to increase the diversity of vegetation fire ages available, break up fuel loads and protect areas of economic and cultural importance (Latz & Green 1995). Regular burning around areas of economic and cultural importance (such as buildings and sacred sites) reduces fuel loads and can therefore reduce their chance of destruction during larger wildfires (Penman et al. 2011). Burning for the benefit of environmental factors is less straightforward. Variations in environmental conditions create large variation in the effect of fire in and between areas (Pastro et al. 2011). In order to implement positive environmental management using fire a clearer understanding of the effect of current fire regimes on population dynamics, survival, and colonization of affected flora and fauna is essential (Sousa 1984; Whelan 1995).

The broad scale control of feral cats has been described as the ‘single most beneficial action’ that could aid the persistence of land mammals in Australia (Woinarski et al. 2014). Although small scale removal (Johnston et al. 2012; Moseby et al. 2009a) or exclusion through fencing (Australian Wildlife Conservancy [AWC] 2015; de Tores & Marlow 2012; Hayward et al. 2014) of both cats and foxes is conducted at many locations in arid Australia, widespread long term removal is currently not feasible. However, research into more effective methods of controlling introduced predators continues (Hsee & Ruan 2015; Read et al. 2015a).

The control of introduced herbivores has been more successful. The release of rabbit haemorrhagic disease (RHD virus) has resulted in an initial 60% decrease of rabbits across Australia in the 1990s, and continues to affect populations
today (Schwensow et al. 2014). More recently extensive efforts have been undertaken to reduce the number of feral camels across Australia through targeted culls with specific recommendations for continued control into the future (Hart & Edwards 2016).

1.5.1. **Monitoring species to assess the effectiveness of management**

There is no established national approach to biodiversity monitoring in Australia. This lack of a national approach has exacerbated the challenges of monitoring biodiversity in the vast, remote, and sparsely populated inland of Australia. In arid Australia cost effective monitoring and management is extremely difficult to maintain over large areas and/or across long enough time frames. To date the majority of ecological based monitoring occurs in the National Reserve System (NRS; network of public, indigenous and private protected areas) which only makes up 22% of arid zone Australia, whilst areas of grazing cover 65% (Department of Environment and Energy 2015). Overall, the level of monitoring across both land tenures is poor, making regional progress difficult.

Although broad scale surveys can assess ecosystem health at a landscape level at relatively low costs, they potentially fail to detect or monitor changes in small populations or cryptic species. In order to detect changes at finer scales, land managers can carry out targeted surveys on key species, which are particularly responsive to changes in the ecosystem and/or are experiencing decline. Although considerably more costly, targeted survey may detect changes before the effect is heavily felt on a broader scale (Landres et al. 1988).

Long term and large fluctuations of species abundances in Australia’s arid zone have been well documented (e.g. Dickman et al. 2010; Letnic & Dickman 2006;
Read et al. 2012). However, little is still known about the factors that influence species persistence during periods when pressures are most intense. Conducting targeted intensive surveys during these periods is essential to gaining this understanding as low capture rates often inhibit the ability to discern patterns from broad scale studies. Optimising the output of these studies is further complicated as the efficacy of different monitoring techniques is highly variable between species and groups. Developing management strategies that incorporate suitable monitoring techniques for common, easily detected species as well as rare, cryptic, and/or patchy species is complicated and often overlooked in management at a landscape scale. By timing the collection of information to periods where pressures are at a peak, I can help to narrow down the key components of the system that promote survival when species are at their most vulnerable. This will help to focus conservation and management efforts to areas with the greatest potential for impact and improve outcomes.

1.6. Project aims

My study investigated various approaches to monitoring the persistence of vertebrate species in spinifex sand plain habitat in arid Australia. I carried out and examined the results of an intensive broad scale repeated trapping design to predict the persistence of readily detected small mammals, reptiles and invertebrates during the initial period of bust. Focusing on a species that is known in persist in the area but was not detected by broader scale monitoring, *Dasycercus blythi*, I conducted a more detailed examination of their occurrence over varying environmental conditions and habitat selection at a fine scale. This provided greater insight into the factors that promote their persistence during periods when pressures are highest. I further assessed how variations in
methodologies can ultimately alter our ability to detect and therefore predict the occurrence of species in the landscape.

The specific aims of the work are as detailed below:

**Aim 1**: To determine what factors are most influential in predicting the persistence of readily detected species, particularly in relation to previous management strategies that focussed on fire and introduced predators (Chapter 3 and 4).

**Aim 2**: To examine where *Dasycercus blythi* are detected during periods of low resource availability to determine the spatial patterns of persistence that could aid in focusing management and monitoring across a broader landscape (Chapter 5).

**Aim 3**: To explore the use of habitat by *D. blythi* at a fine scale in relation to foraging strategies and predator exposure and identify the characteristics that help support *D. blythi* populations (Chapter 6).

**Aim 4**: To explore and compare the benefits and costs of several monitoring methods and their ability to detect species (Chapter 7).

**Aim 5**: To collate the findings from these studies to develop recommendations to further refine management strategies for spinifex sand plains, particularly in relation to *D. blythi* (Chapter 8).
CHAPTER 2

STUDY SITES, SPECIES ASSEMBLAGES
AND RECENT HISTORY

Clockwise: Sunrise at Newhaven Wildlife Sanctuary (J Molyneux); Sunrise at the Borefields (J Molyneux); prescribed burning (J Schofield/AWC); *Canis dingo* at Newhaven Wildlife Sanctuary (J Molyneux)
2.1. Study sites

2.1.1. Location

The study was conducted across two sites; Newhaven Wildlife Sanctuary and the Yulara/Uluru borefields (Figure 2.1). Newhaven Wildlife Sanctuary (henceforth referred to as Newhaven) is 363 km north-west of Alice Springs, located on the eastern boundary of the Great Sandy Desert Bioregion (22°43’S, 131°10’E). Newhaven covers 262,000 hectares and is situated close to the boundary of both the Burt Plain and MacDonnell Ranges Bioregions. The property is surrounded by Aboriginal freehold land held by two land trusts (Ngalurrtju and Yunkanjini).

The second study site focused on the borefields area (henceforth referred to as ‘the Borefields’), which crossed the boundary between Ayers Rock Resort and Uluru Kata-Tjuta National Park, some 346 km south-west of Alice Springs and on the south-eastern tip of the Great Sandy Desert bioregion (25°16’S, 130°58’E). The Borefields area occurs from the south east of the Ayers Rock Resort leased lands into the central parts of the Uluru Kata-Tjuta National Park. Study sites were situated on extensive paleo drainage lines and were approximately 280 km apart.
2.1.2. Climate

Both study sites are considered hot deserts (Williams & Calaby 1985) and are heavily influenced by the El Niño Southern Oscillation (ENSO). The ENSO is a naturally occurring cycle in the Southern Hemisphere between two extreme events; La Niña (periods of high rainfall) and El Niño (periods of low rainfall; Nichols 1991). Both events can occur over relatively short time periods creating large fluctuations in rainfall between consecutive years (Nichols 1991). In relation to my research period, the last La Niña event began in 2010 and finished in 2012 which included two years of high rainfall, particularly over summer (Bureau of Meteorology [BOM] 2011). Following this, rainfall declined sharply, officially transitioning into the initial stage of an El Niño event in February 2014 (BOM 2015a). Although both sites are heavily influenced by these fluctuations, site-specific climate variations are also apparent.
Since 1965, mean annual temperatures at Newhaven ranged between 21 – 25°C with winter minimums to -2 °C and summer maximums to 44 °C (BOM 2015b). Rainfall is variable in the region with the lowest annual rainfall recorded at Newhaven being 66 mm in 1963 and the highest 951 mm in 1974 (BOM 2015b). For the purpose of this study, I gathered data daily for total rainfall and maximum and minimum temperatures. In February 2013, a Davis Vantage Vue Weather Station (Instrument Choice, Regency Park, SA, Australia) was installed on-site to record temperature fluctuations. Prior to February 2013, I utilised the temperature data from the closest weather station. Due to inconsistent data collection by surrounding weather stations this resulted in 55%, 35% and 10% of temperature data being gathered from weather stations at Yuendumu (83 km north-east from site), Papunya (93 km south-east from site) and Kintore (194 km south-west from site), respectively. Daily rainfall was collected from Newhaven homestead (between 4 and 29 km from survey sites) for the entire duration of the study. Although the region moved from a transitional to an El Niño phase during the study, Newhaven experienced two high rainfall events in May 2013 (122mm) and January 2014 (189.5mm; Figure 2.2). Although these are considered unusual at a regional scale rainfall events of similar size have regularly been recorded at the property during previous El Niño phases (BOM 2015b) due to the monsoonal influences from the north-west (henceforth I have referred to these events as “top up” rainfall).
Figure 2.2 Climate statistics recorded at Newhaven Wildlife Sanctuary prior and during study for (a) yearly long term rainfall since 1996 and (b) short term monthly rainfall (mm) and temperature variables (°C) since May 2011. ♦ indicates survey periods. Rainfall values are given on the primary axis (left); black bars indicate total monthly rainfall. Temperature values are given on the secondary (right) axis; Dark grey line indicates average maximum temperature and light grey line indicates average minimum temperature. NB: Nov-14 totals end on 15/11/2014 to reflect last day of surveying.

Since 1983, the mean annual temperature at the Borefields has ranged between 14 – 29 °C. Temperatures fluctuate from winter minimums of -3 °C to summer maximums of 47 °C. As expected, annual rainfall varies considerably with the lowest annual rainfall recorded being 126.9 mm in 1985 and the
highest 825.4 mm in 2001 (BOM 2015c). As with Newhaven, I gathered data daily for rainfall and minimum temperatures. All climate data were collected from a weather station at Yulara Airport (10km north-east from site; BOM 2015c). All rainfall totals were considered representative of the transitional state into El Niño (Figure 2.3).

Figure 2.3 Climate statistics recorded at Borefields prior and during study for (a) yearly long term rainfall since 1996 and (b) short term monthly rainfall (mm) and temperature variables (°C) since October 2012. ♦ indicate survey periods. Rainfall values are given on the primary axis (left); black bars indicate total monthly rainfall. Temperature values are stated on the secondary (right) axis; Dark grey line indicates average maximum temperature and light grey line indicates average minimum temperature. NB: Jan-15 totals end on 16/1/2015 to reflect last day of surveying.
2.1.3. Vegetation assemblages

The study areas in both Newhaven and the Borefields are classified as spinifex sand plain/dunes habitats (Buckley 1981; Latz et al. 2003). At both sites, vegetation is dominated by the hummock grass, Triodia (commonly referred to as spinifex), with intermittent, low-density pockets of low shrubs and trees. Site-specific differences are apparent at a species level with *T. pungens* dominating in the central areas of Newhaven and *T. basedowii* dominating across the area surrounding this. Large areas of dune fields are apparent in the southern regions and surrounding the main ephemeral lake, Lake Bennett (Figure 2.4a). In contrast, in the UKJNP and Ayres rock resort *T. basedowii* dominates the northern and central borefields area whilst *T. pungens* dominates across the southern and eastern areas of the park (Figure 2.4b). Both areas in UKJNP and Ayers Rock Resort display intermittent dunes across their extent.
Chapter 2 | Study Sites

Figure 2.4 Vegetation classification in (a) Newhaven Wildlife Sanctuary and (b) UKJNP and Ayres Rock Resort. Red indicates extent of *Triodia basedowii* plains; Orange indicates extent of *Triodia pungens* plains and pink indicates dune field areas; Blue indicates ephemeral lake systems; White indicates extent of non-*Triodia* dominant habitats.

The biology of the two dominant spinifex species differs considerably as *T. pungens* is variable in form, can spread through seed and runners and is quick to re-establish after fire (Latz 2007). In comparison *T. basedowii* is quite distinctive in form, tending to die out in the middle to form rings (Latz 2007). It spreads through seed dispersal and is slower growing, taking at least 5 – 10 years (depending on rainfall) to re-establish after fire (Casson & Fox 1987).

Intermittent shrub layers are apparent at both sites. At Newhaven *Melaleuca glomerata* and *Grevillea juncifolia* dominated (Latz et al. 2003) whilst...
*Eremophila* species and *Allocasuarina decaiseana* dominated at the Borefields (Buckley 1981). Forbs and other small grasses also become prevalent for a short periods immediately following rainfall in recently burnt areas.

### 2.2. Vertebrate assemblages

Although Australia’s arid zone is considered nutritionally poor due to the low nutrient levels in soils and unpredictable rainfall, it still supports relatively high vertebrate (Stafford Smith & Morton 1990) and invertebrate diversity (Kwok & Eldridge 2015; Landsberg et al. 1999; Morton et al. 2011). Information available on vertebrate assemblages is more detailed than that for invertebrates at both sites. Here I outline available knowledge on the assemblages of small mammals and reptiles in the spinifex sand plains habitat at each site.

#### 2.2.1. Mammals

Both study sites are inhabited by similar mammal assemblages (Table 2.1; AWC 2014; Balding 1996). A total of 17 native and eight introduced species of mammals have been identified as occurring or potentially occurring in the study sites. Although up to 15 bat species occur in the southern Northern Territory and most are known to roost in tree hollows (Milne & Pavey 2011), they are not included here as I did not sample in this group during this study. Throughout this study I refer to canines as “wild dogs” as our detection methods could not distinguish between specific canine species. Notably, property managers, Australian Wildlife Conservancy, classify most canids on the property as dingoes (*Canis dingo*; Kanowski J, pers. comm.).
Table 2.1 Terrestrial mammal species recorded or potentially occurring in the spinifex sand plain habitat at Newhaven Wildlife Sanctuary (NH) and the Borefields (BF). ✓ indicate confirmed occurrence through trapping records or observation at the site; ? indicate potentially could occur at the site (Van Dyck et al. 2008).

<table>
<thead>
<tr>
<th>Order/Family</th>
<th>Scientific Name</th>
<th>Common Name</th>
<th>NH</th>
<th>BF</th>
</tr>
</thead>
<tbody>
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<td>Dingo</td>
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<td>✓</td>
</tr>
<tr>
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<td>Kultarr</td>
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<td>?</td>
</tr>
<tr>
<td></td>
<td><em>Dasyurus blythi</em></td>
<td>Brush-tailed mulgara</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Ningaui ridei</em></td>
<td>Wongai ningaui</td>
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<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Sminthopsis crassicaudata</em></td>
<td>Fat-tailed dunnart</td>
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</tr>
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<td><em>S. hirtipes</em></td>
<td>Hairy-footed dunnart</td>
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</tr>
<tr>
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<td>Striped-faced dunnart</td>
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</tr>
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<td>Ooldea dunnart</td>
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</tr>
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<td>Monotremata</td>
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</tr>
<tr>
<td>Notoryctoidae</td>
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<td>Southern marsupial mole</td>
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<td>✓</td>
</tr>
<tr>
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<td>Central short-tailed mouse</td>
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</tr>
<tr>
<td></td>
<td><em>Notomys alexis</em></td>
<td>Spinifex hopping-mouse</td>
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<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Pseudomys desertor</em></td>
<td>Desert mouse</td>
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<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>P. hermannsburgensis</em></td>
<td>Sandy inland mouse</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Rattus villosissimus</em></td>
<td>Long-haired rat</td>
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<td>✓</td>
</tr>
<tr>
<td>Introduced</td>
<td><em>Felis catus</em></td>
<td>Cat</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>species</td>
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<td>Rabbit</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Camelus dromedarius</em></td>
<td>Camel</td>
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<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Mus musculus</em></td>
<td>House mouse</td>
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<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Vulpes vulpes</em></td>
<td>Red fox</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Bos taurus</em></td>
<td>Cattle</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Equus ferus</em></td>
<td>Horse</td>
<td>?</td>
<td>?</td>
</tr>
<tr>
<td></td>
<td><em>E. asinus</em></td>
<td>Donkey</td>
<td>?</td>
<td>?</td>
</tr>
</tbody>
</table>
2.2.1.1. **Factors influencing small mammal persistence**

At both sites the abundance and distribution of arid zone small mammals is strongly influenced by La Niña events and the subsequent increased primary productivity and resources (Schwinning et al. 2004). Facilitated by increases in productivity, rodents are capable of reproducing rapidly resulting in large population irruptions (Dickman et al. 1999; Greenville et al. 2012; Predavec 1994; Southgate & Masters 1996). Most dasyurids do not have this flexibility in breeding strategies and therefore do not experience the large fluctuations seen in rodents (Dickman et al. 2001; Greenville et al. 2012; Letnic & Dickman 2010). Instead, many survive through behavioural adaptations that exploit periodic patchy increases in productivity or enable persistence during periods of low resources. These adaptations include ‘drifting’ home ranges that move to resource rich patches as they become available (Bos & Carthew 2007a; Haythornthwaite & Dickman 2006b) or specific energy saving behaviours, such as daily torpor (Körntner et al. 2008; Leslie et al. 2015; Pavey et al. 2009).

Following La Niña events many species of small mammals decrease rapidly in abundance and/or distribution due to reducing resource availability, and/or high predation (Letnic & Dickman 2006; Letnic & Dickman 2010; Pavey et al. 2014). It is during this period of intense ecological pressure that our study was conducted.

2.2.2. **Reptiles**

The spinifex grasslands in Australia are considered to support the world’s richest lizard assemblage (Morton & James 1988; Pianka 1969b). It is not unsurprising therefore that a total of 67 reptile species have been identified from the two areas, including 55 species of lizard (six dragon, 12 gecko, 27 skink, four monitor, six legless lizard) and 12 snake species. Most species occur across
both study sites; however, the known distributions of 10 species restricts their occurrence to only one of the locations studied (Table 2.2; AWC 2014; Balding 1996; Wilson & Swan 2013).

Table 2.2 Reptile species recorded or potentially occurring in the spinifex sand plain habitat Newhaven Wildlife Sanctuary (NH) and the Borefields (BF). ✓ indicates confirmed occurrence through trapping or observation at the site; ? indicates potentially could occur at the site (Wilson & Swan 2013).

<table>
<thead>
<tr>
<th>Order/Family</th>
<th>Scientific Name</th>
<th>Common Name</th>
<th>NH</th>
<th>BF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agamidae</td>
<td><em>Ctenophorus isolepis</em></td>
<td>Central military dragon</td>
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<td>✓</td>
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<tr>
<td></td>
<td><em>C. nuchalis</em></td>
<td>Central netted dragon</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>C. clayi</em></td>
<td>Black-collared dragon</td>
<td>?</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>C. pictus</em></td>
<td>Painted dragon</td>
<td>✓</td>
<td>?</td>
</tr>
<tr>
<td></td>
<td><em>Pogona vitticeps</em></td>
<td>Central bearded dragon</td>
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<td>✓</td>
</tr>
<tr>
<td></td>
<td><em>Moloch horridus</em></td>
<td>Thorny devil</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Gekkonidae</td>
<td><em>Diplodactylus conspicillatus</em></td>
<td>Fat-tailed gecko</td>
<td>✓</td>
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2.2.2.1. Factors influencing reptile persistence

The diversity of reptiles at both sites is generally thought to be a consequence of the diversity in habitats created by the dynamic and conflicting effects of fire and rainfall (Nimmo et al. 2012; Pianka 1989). The habitat accommodation model (Fox 1982) has previously been used to explain habitat selection of reptiles in other environments. This model describes reptile occurrence being dictated by specific requirements for vegetation and cover, that are filled for short periods during regrowth stages after fire (Pianka & Goodyear 2012). Recent studies at other arid sites have shown more complex relationships which include food availability (Pastro et al. 2013) and temperature (Read et al. 2012) as important drivers of reptile occurrence and diversity.

2.3. Recent history of species assemblages and land management

Arid Australia has experienced some of the most globally significant extinctions and declines in small to medium sized mammals in the last century (Johnson 2006). The primary driver of declines is thought to be predation by the introduced predators, the feral house cat, *Felis catus*, and the European red fox, *Vulpes vulpes* (Woinarski et al. 2014); however, several secondary influences have also been identified including degradation of habitat, competition from introduced herbivores and changes in land management, particularly in relation to fire (Woinarski et al. 2007).

Recently (in the last 150 years) a dramatic change in the predator dynamics has occurred through the expansion of the feral cat and red fox into Australia’s arid zone (Spencer et al. 2014). Both these predators have greatly impacted the trophic structures of the region and have been heavily implicated in the decline and extinction of a large number of species (Johnson 2006). The presence of
introduced predators is still having a dramatic effect on ecosystem dynamics, particularly after periods of high rainfall, when predator abundance is high and prey availability is becoming low (Greenville et al. 2014). During these periods, top-down pressure from introduced predators is considered a primary concern for many species in the arid zone.

During the periods following rainfall, fire is also considered a major factor influencing the persistence of many arid zone species (Letnic et al. 2004; Letnic et al. 2005; Pastro et al. 2011; Pianka & Goodyear 2012). As rainfall decreases, vegetation dries out creating high fuel loads that can result in large-scale wildfires (Greenville et al. 2009). Many of Australia’s arid zone flora and fauna show relatively high tolerances to fire due to a long history of both lightning and human ignited fires (Bradstock et al. 2002). Historically, indigenous people utilised fire to manage the landscape for food, although these fires were typically of lower intensity and smaller than the wildfires experienced after extreme rainfall events (Bird et al. 2012). After European colonisation the use of fire by humans reduced dramatically, apparently altering many of the ecological relationships developed over thousands of years (Bradstock et al. 2002). In recent times, the re-implementation of active human induced fire management has increased. Currently many land managers utilise “prescribed burning” to manage large remote landscapes (Pastro et al. 2011; Schultz 2006; Smith et al. 2013). The impact of large-scale fires is still being felt, particularly in the years following high rainfall. Many species of fauna showing a strong negative response caused by increased exposure to predators and resource removal (Letnic & Dickman 2005; Schultz 2006; Woodrow 1992). The last major wildfire event that occurred across arid Australia prior to this study was in 2011 and 2012 when approximately 25% and 55% of UKTNP/Ayers Rock Resort and
Newhaven was burnt, respectively. This occurred despite intensive efforts to minimise the extent of wildfires across both properties through fire management regimes.

The presence or absence of fires can severely alter suitability of habitats depending on the species of interest. Furthermore, hyper predation during drought, partially brought about by the increased availability of water across the broader landscape is thought to have contributed strongly to the demise of many small mammal species in the arid zone, particularly during dry periods when ecosystem stressors are high (Dickman 1996a; Dickman et al. 1993; Letnic & Dickman 2005; Mahon 1999).

2.3.1. Management at Newhaven Wildlife Sanctuary
Newhaven Wildlife Sanctuary (previously known as Newhaven Station) was managed as a pastoral cattle-grazing property from 1959 until late 2000 when it was purchased by Birds Australia (now Birdlife Australia) as a conservation reserve (Fitzherbert 2000). As a pastoral lease, the property had 12 active bores and was conservatively managed with 25% of the area never grazed and the remaining 75% only lightly stocked (Fitzherbert 2000). Between 1997 and 2005 the property was subject to extensive fires instigating the creation of the Newhaven Fire Management Strategy which detailed the implementation of a five year plan beginning in June 2006 (Paltridge & Latz 2006). This plan recommends the use of human induced fire to create fire breaks and areas of varying fire history, particularly in the spinifex sand plains habitat (Paltridge & Latz 2006). In 2006, Birdlife Australia partnered with Australian Wildlife Conservancy (AWC) to jointly manage the property, with AWC becoming the lead managing partner (Fitzherbert 2000). In April 2008, AWC instigated annual fauna surveys at 75 sites stratified across all habitats types represented
in Newhaven (Latz et al. 2003) and annual tracking surveys to monitor large mammalian predators (particular introduced species) and native vertebrates. In recent years, management focus has continued to focus on fire management and annual monitoring. An increased emphasis on predator control through fox baiting in 2014 and continual weed removal around key areas of concern has been seen since 2006 (AWC 2016b). Currently four bores remain active across the property (J Schofield 2016, pers. Comm).

2.3.2. Management at the Borefields
The Borefields area is currently managed across two landholders; Ayers Rock Resort and the Uluru Kata-Tjuta National Park (UKTNP). The area has been a focal tourism destination since the 1950s. However, the tourism township, Yulara, was not established until the 1970s (Voyages Ayers Rock Resort [VARR] 2016). In 1997, the area surrounding Yulara (now known as Ayers Rock Resort; see Figure 2.1) was privatised and has since been managed under several organisations (currently Indigenous Land Corporation; VARR 2016). The Yulara township has expanded considerably since then, currently supporting over 1000 permanent residents and supplying a variety of accommodation options and services to the area (VARR 2016). Over the course of its expansion, the resort has strived to minimise the environmental impact and delivered some on-ground environmental management in the form of periodic culling of introduced predators and vertebrate monitoring in the Borefields area (R Paltridge & C Watson 2014, pers. Comm).

In contrast UKTNP has been managed relatively intensively since its formation in 1977 as an important conservation and cultural region (Australian Director of National Parks [ADNP] 2010). In 1985 park ownership was handed back to its traditional aboriginal owners, Anangu, and in turn
leased back to the Director of National Parks (ADNP 2010). Since then the area has been jointly managed through the Uluru Board of Management with the park being granted World Heritage listing in 1987 (ADNP 2010). Due to the large diversity of interested parties in the property, management is broad including considerations for cultural, tourism, ecological and economical aspects. A 10-year management plan was created in 2010. Ecological programs focused on regular fauna monitoring, maintenance of a introduced predator free enclosure for threatened or extinct species in the region, removal of introduced plants and animals (particularly buffel grass, camels, rabbits and cats) and the implementation of a fuel reduction burning program (ADNP 2010).
CHAPTER 3

SPECIES OCCURRENCE IN A POST BOOM PERIOD

PART A: SMALL MAMMALS AND REPTILES

Clockwise: Delma tincta (J Molyneux); Diplodactylus conspicillatus (A Molyneux); Dasycercus blythi (A Molyneux); Pseudomys desertor (J Molyneux).
Abstract
The abundance of Australia’s arid zone vertebrates is typically driven by large fluctuations in rainfall, which dictate productivity across functional groups. Many species fluctuate from highly abundant in high rainfall periods to low abundance or even local extinction during low rainfall periods. These large-scale fluctuations are well documented; however, much less is known regarding the factors that influence persistence of fauna when pressures are greatest (periods of low rainfall). This study examined the influence of a variety of biotic and abiotic factors on the richness and abundance of nocturnal small mammals and reptiles in the spinifex sand plains of central Australia. The study was conducted during the initial period of bust, immediately following the dramatic decline of most vertebrate species in the area. Mammals showed species-specific responses, with varying dependencies on cover and climatic factors, whilst reptiles showed a surprisingly limited response to resource abundance and rainfall. The most consistent predictor for both small mammals and reptiles was the presence of large mammalian predators (cat and wild dog) across the area. An inverse relationship between the presences of the two predators indicated that cats had greater suppressive effects on small mammals whilst reptiles declined with greater wild dog presence. These results follow the patterns seen in predator diets in the region and likely represent the selective foraging of the two predators in the landscape. My results highlight the importance of implementing predator control during periods of low rainfall to reduce the potential suppressive effects larger predators may have on native vertebrates.
3.1. **Introduction**

Understanding how species interact with their environment and how these relationships change over time is one of the most complex and difficult challenges faced by ecologists (Cornell & Lawton 1992). By understanding these processes, land managers can develop more efficient and targeted management strategies that support individual species and the overall health of an ecosystem (Holling 2001). However, gaining this understanding can often be challenging as it relies heavily on the ability to readily detect and monitor species across time and space. In Australian arid regions, readily detecting the presence and relative abundance of fauna species can be particularly difficult as abundance varies considerably over time. This variation flows from periods of high rainfall (boom) where animals increase in numbers and can be detected relatively easily to periods of low rainfall (bust) where species decline or even disappear from areas (Dickman et al. 2001; Dickman et al. 1999; Letnic & Dickman 2006; Letnic et al. 2005).

During times of low rainfall following booms, bottom-up pressures on the ecosystem increase as resources become limited and reduce the systems’ ability to sustain large abundances (Greenville et al. 2014). The flammability of vegetation and the accumulation of large fuel loads during high rainfall periods can also result in extensive fires in the post boom period (Orians & Milewski 2007). Due to the expansion of cats and foxes into the arid zone, predatory pressure, particularly during periods post boom, has increased dramatically (Dickman 1996a). Changes in land management have also altered the landscape through the removal of vegetation and the facilitation and introduction of dominant consumers, such as livestock, camels and rabbits (Landsberg et al. 1997). This not only has impact on populations through direct
mortality but also indirectly through further removal of resources and creation of open areas where exposure to predators is higher (Körtner et al. 2007; Letnic et al. 2013). Gaining information on how the myriad of factors influence persistence during a bust period is clearly complex, but critical to ensure we manage ecosystems effectively during times when species are most vulnerable.

The factors influencing the occurrence of fauna in Australia’s arid zone are relatively well studied. Fire is considered highly influential although responses vary among species. Some, such as *Pseudomys desertor*, clearly favouring unburnt areas where cover is higher (McDonald et al. 2016) whilst other species, such as *Macrotis lagotis*, are thought to actively select areas in close proximity to recently burnt habitat to feed on regenerating vegetation (Southgate et al. 2007). Regardless, fire is thought to have a limited impact on most small to medium sized mammal species persisting in Australia’s arid zone when compared to other influences in the system, such as introduced predators (Letnic et al. 2004; Letnic et al. 2005; Letnic et al. 2013). In contrast, many studies have shown fire to influence the occurrence of reptiles through the habitat accommodation model. This model describes how the loss and subsequent regeneration of vegetation dictates the successional stages of reptile occurrence (Fox 1982; Letnic et al. 2004; Masters 1996; Pianka 1996). Climate variables, particularly rainfall, are known to affect fauna through boosting primary production, growth and increasing the availability of food and cover (Doherty et al. 2015a; Letnic et al. 2004; Letnic et al. 2005). In contrast, predation, particularly by introduced predators, is considered one of the primary drivers of decline in arid zone vertebrates (Dickman 1996a). Previous studies on introduced predator diets have shown they are capable of consuming a variety of taxa and capitalise on abundant prey when available (Kutt 2011; Paltridge...
In particular, cats are predominantly thought to be the primary suppressors of small mammals (Risbey et al. 2000; Spencer et al. 2014). Although not an introduced species, dingoes are considered an important predatory regulator in the system displaying preferences for larger mammals and sometimes reptiles (Paltridge 2002; Pavey et al. 2008; Spencer et al. 2014). Most of the studies examining these interactions have focused on one or two factors (i.e. fire, fire and predators, or rainfall and vegetation) and therefore do not assess how many factors interact simultaneously with the abundance of small mammals and reptiles. Furthermore, recent studies have shown a large variability in mammal and reptile responses and have cautioned against the generalisation of information across temporal and spatial scales (Letnic et al. 2011; Pastro et al. 2013; Read et al. 2012). These studies have also highlighted the potential influences of a range of factors including fire, predation, resource availability, climate statistics (i.e. rainfall and temperature) and vegetation that interact to create a dynamic ecosystem that fluctuates across time and space (Letnic et al. 2011).

In this study I focused on the early stages of bust (Figure 2.2 and Figure 2.3), to assess which factors were the most important predictors of persistence when species are most vulnerable. I examined factors previously identified as drivers of small mammal and reptile occurrence, including rainfall, temperature, vegetation cover, resource availability, fire and predation. By including all variables simultaneously, I could identify the most dominant factors influencing persistence whilst also incorporating the influence of less important factors. In particular, I predicted that:

1. Variation in vegetation cover will have a limited effect on most species with the exception of reptiles and mammals known to prefer areas with
greater cover (i.e. *Ctenotus* spp. and *P. desertor*) which will prefer long unburnt areas (>25 years).

2. Greater rainfall will promote an immediate increase in productivity, species richness and abundance.

3. Temperature will alter mammal activity and therefore capture rates to reflect species-specific physiological requirements (i.e. breeding conditions), whilst reptiles will respond more uniformly to seasonal fluctuations.

4. Large mammalian predators will have a considerable negative impact on the abundance of native vertebrates. In particular, cats will have a strong impact on small mammal abundance whilst to a lesser degree wild dogs will influence reptiles.

5. Fluctuations in invertebrates will have a considerable effect on abundance and activity of insectivorous reptile and mammal species.

### 3.2. Methods

#### 3.2.1. Study Site

Newhaven Wildlife Sanctuary is 363 km north-west of Alice Springs, located on the eastern boundary of the Great Sandy Desert Bioregion. The area is considered semi-arid with a mean annual rainfall of around 334.2 mm.

The landscape is predominantly spinifex sand plains habitat, which was the focal habitat of this study. The sand plains are dominated by the spinifex species *Triodia pungens*, with intermittent, low-density pockets of low shrubs. Common shrubs in the area include *Melaleuca glomerata* and *Grevillea juncifolia*. Forbs and other small grasses also become prevalent immediately following rainfall in recently burnt areas (See Chapter 2 for detailed description of the study site).
This study was conducted between November 2012 and November 2014. Annual rainfall increased from 187.6 mm in 2012 to 381.4 mm in 2014 (BOM 2015b). During the survey period higher than expected rainfall was recorded in May 2013 (122 mm) and January 2014 (189.5 mm).

3.2.2. Small mammal and reptile sampling

Twelve sites were selected in a 30 km by 5 km band of spinifex sand plains habitat at Newhaven (Figure 3.1). Following the design of previous studies in similar habitats (Dickman et al. 2011; Masters 1993, 1996) neighbouring sites were a minimum of 1.5 km apart.

![Figure 3.1. Site locations across Newhaven Wildlife Sanctuary. Shaded area indicates extent of spinifex sand plains habitat.](image)

Each site was surveyed quarterly for small mammals and reptiles from November 2012 to November 2014 (total of nine survey periods). At each site, four pitfall traps and 25 box traps were deployed for three consecutive nights. Aluminium box traps (type A: 30 x 10 x 8 cm; Elliott Scientific Equipment,
Upwey, VIC, Australia) were set in 5 x 5 grid with 10 metres between individual traps. Pitfall traps were placed in pairs running diagonally through the grid. Each pitfall line consisted of two PVC pipes (16 cm diameter, 60 cm deep), buried flush to the ground, and situated approximately 15 metres apart. A dampcourse drift fence (30 cm high) ran between each pair of pitfalls and for a further 3 - 5 metres on either side. PVC lids were glued to the bottom of pipes to prevent animals from escaping. Between trapping surveys, all box traps and pitfall drift fences were removed whilst pitfall pits were capped with lids and mounded with sand. During surveys box and pitfall traps checked early morning, closed during the day and opened late afternoon over three consecutive days. All bait was removed during the day and replaced each night to reduce ant activity around each trap site. All individuals caught were classified to species level. To assess the overall level of diversity in nocturnal small mammals and reptiles I calculated each site’s overall mammal and reptile richness (# species captures) at each survey period. To further gauge the response of individual species, I calculated individual abundances in each survey period (# of individuals captures) for species with sufficient captures (over 30 individual captures and greater than 15% site detection rate across all surveys).

3.2.3. Invertebrate sampling
At each site four large (16 cm diameter, 60 cm deep) and four small (75 mm diameter, 130 mm deep) pitfalls were used to sample invertebrates during all mammal and reptile surveys (for sampling protocols see Chapter 4). Only ground dwelling invertebrates were sampled as they represented the largest component of the diets recorded for small mammals and reptiles in the area (Bos & Carthew 2007b; Chen et al. 1998; Fisher & Dickman 1993a, 1993b; Twigg et al. 1996). All invertebrates caught were identified to order and classified into
five size classes (0 - 5 mm, 6 - 10 mm, 11 - 15 mm, 16 - 20 mm and > 20 mm). A metric for invertebrate biomass was then calculated by multiplying abundance by median size (i.e. 7.5 mm for size class 6 – 10 mm; henceforth-referred to as invertebrate index). Due to the prolific abundance of Hymenoptera in samples (88.49% of invertebrates sampled) they were recorded separately from all other orders.

3.2.4. Fire data
Locations of trapping sites in the spinifex sand plains habitat were selected based upon a combination of accessibility and recent fire history. I considered only fires since 2009 as high rainfall in 2010 and 2011 resulted in considerable recovery of burnt vegetation and fire scars prior to 2009 were no longer clearly distinguishable (verified by on-ground surveys). I assessed fire extent across two spatial scales; fine scale (the area covered by the trapping grid) and broad scale (the 40 hectares surrounding each grid). The area of 40 hectares was used as this sampled the available habitat surrounding each individual grid without overlapping neighbouring grids. At a fine scale, sites were classified as burnt or unburnt since 2009. Over the same period, I also classified each site across the broader 40 ha scale into one of three categories; unburnt (0 - 15%), partially (patchy) burnt (15 - 85% burnt) and completely burnt (> 85% burnt). At both fine and broad scales each category was equally represented in the sites (i.e. six sites represented each fine scale category and four sites represented each broad scale category).

3.2.5. Habitat structure data
To determine changes in habitat structure over the course of sampling I assessed vegetation/ground cover at each site during each of the nine survey periods. I used a point intercept method along a transect that ran diagonally
across the trapping grid to record the dominant vegetation or substrate type at each 1 metre point. If vegetation was present at a point it was classified into *Triodia*, other grasses, forbs, shrubs and log. If no vegetation was apparent then the dominant ground cover was recorded within a 20 cm radius of the point. Ground cover was classified into bare ground, leaf litter and termite mound. To ensure that vegetation surveys represented the habitat available at a broader scale I compared the results to identical surveys conducted annually at four randomly selected locations within the broader 40 hectares. No difference was found between surveys conducted at trapping grids and at broader random locations, therefore data from trapping grids was used for all statistical analyses.

3.2.6. **Large mammalian predator data**

To assess large mammalian predator presence at each site, a 30 minute search in the immediate area surrounding each trapping grid (maximum 200 m from grid) was conducted during each survey period to detect recent sign of cat, fox and/or wild dog (i.e. fresh/ non-dried scats or tracks). Sign of recent predator presence was removed after being recorded as either present or absent, giving an index for each predator species during each survey at each site. Based upon previous studies on predator movement (e.g. Moseby et al. 2009a; Thomson 1992) sites were not deemed spatially independent (1.5 – 28 kms between sites) therefore data across all sites were collated to provide the proportion of sites with recent activity for each predator during each survey period. Due to the low number of sites where foxes (n = 2) were recorded across all surveys they were not included in statistical analyses.
3.2.7. **Climate data**
For 18 months prior to sampling and throughout the study, rainfall and minimum and maximum temperature were collected daily (for details on collection of climate data see Chapter 2). I examined three rainfall variables; total rainfall, average rainfall event size and the number of rain days, and two temperature variables; average daily maximum temperature and minimum temperature. All variables were calculated accumulatively for each day up to 18 days prior to surveys to represent ‘short term’ climate and each month up to 18 months prior to surveys to represent ‘long term’ climate.

3.2.8. **Statistical analysis**
I first explored potential inter-species relationships through non-parametric Spearman’s rank-order correlation coefficient. This provided information on the linear relationship between species abundances whilst allowing for a lack of normality in the data. Second, I investigated the influence of the various environmental variables on the abundance and richness of small mammal and reptile through generalised additive mixed models (GAMMS; Zuur et al. 2014). I utilised GAMMs to account for the repeated measures trapping design and because initial data exploration showed clear non-linear relationships between response and predictor variables (Zuur et al. 2014). To reduce the number of habitat structure variables in models I conducted principal component analysis (PCA) on all vegetation and ground cover variables. This assessed which contributed to the most site-specific differences. As cumulative climate variables showed large collinearity (Spearman rank-order correlations > 0.60), I used only the variable with the strongest correlation to the response variable in each final model (Table 3.1). Preliminary data exploration showed an expected seasonal variation in reptile abundance caused by their ectothermic requirements. To
account for this known seasonal response I included the average minimum temperature recorded over the 3 days of sampling for all reptile models instead of short and long-term temperature variables. This resulted in two climate variables (two rainfall) and four climate variables (two rainfall and two temperature) being used in reptile and mammal models, respectively (Table 3.1). All continuous covariates were standardized to account for the different measurement scales among covariates (Zuur et al. 2009).

Table 3.1 Climate variables used in generalised additive mixed models for each response variable examined. Numbers in parentheses indicate number of days (short term) or months (long term) included in accumulated totals.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>Temperature</th>
<th>Rainfall</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Short term</td>
<td>Long term</td>
</tr>
<tr>
<td>Mammal Richness</td>
<td>Min. (4)</td>
<td>Max. (11)</td>
</tr>
<tr>
<td>P. desertor</td>
<td>Min. (2)</td>
<td>Min. (7)</td>
</tr>
<tr>
<td>P. herman.</td>
<td>Min. (9)</td>
<td>Max. (11)</td>
</tr>
<tr>
<td>Reptile Richness</td>
<td># days (16)</td>
<td># days (1)</td>
</tr>
<tr>
<td>C. hanloni</td>
<td># days (14)</td>
<td>Ave. event (11)</td>
</tr>
<tr>
<td>C. schomb.</td>
<td># days (14)</td>
<td>Total (13)</td>
</tr>
<tr>
<td>N. ornatus</td>
<td># days (14)</td>
<td>Total (13)</td>
</tr>
<tr>
<td>L. bipes</td>
<td># days (18)</td>
<td># days (14)</td>
</tr>
<tr>
<td>L. labialalis</td>
<td># days (18)</td>
<td># days (1)</td>
</tr>
</tbody>
</table>

All variables were checked for outliers and collinearity using Pearson correlation coefficients (< 0.70) and variance inflation factors (< 3; Zuur et al. 2014). If two variables showed strong correlation, I retained the variable considered most ecologically relevant based upon previous knowledge of the response variable. All models included site as a random factor, to account for repeated sampling. Initially I included log (number of trap nights) as a covariate to account for
variations in trap effort between sites and surveys; however, as estimated parameters were close to one I followed the recommendations of Zuur et al. (2014) and included it as an offset in final models. The offset function assumes a proportional relationship with the response variable meaning greater trap effort resulted in a higher chance of detection.

Due to insufficient replicates in the data and to avoid over-fitting, I was unable to include all variables simultaneously. Therefore, variables were grouped by ecological relevance (Table 3.2). I then created a set of 21 candidate models that comprised all single and pair-wise combinations of the variable groups (including the null model; Appendix 1).

<table>
<thead>
<tr>
<th>Group</th>
<th>Variables included in group</th>
</tr>
</thead>
<tbody>
<tr>
<td>Habitat structure</td>
<td>All vegetation and ground cover variables</td>
</tr>
<tr>
<td>Predation</td>
<td>Cat presence</td>
</tr>
<tr>
<td></td>
<td>Wild dog presence</td>
</tr>
<tr>
<td>Invertebrate</td>
<td>Ant index</td>
</tr>
<tr>
<td></td>
<td>Other invert index</td>
</tr>
<tr>
<td>Interaction</td>
<td>Mammal abundance (Reptile models)</td>
</tr>
<tr>
<td></td>
<td>Reptile abundance (Mammal models)</td>
</tr>
<tr>
<td>Short term climate</td>
<td>Short term temperature (mammal only)</td>
</tr>
<tr>
<td></td>
<td>Short term rainfall</td>
</tr>
<tr>
<td>Long term climate</td>
<td>Long term temperature (mammal only)</td>
</tr>
<tr>
<td></td>
<td>Long term rainfall</td>
</tr>
</tbody>
</table>

Initial data exploration showed obvious zero inflation in the data; therefore, a zero-inflated Poisson (ziP) distribution with an identity link function was used for all models. I ranked all candidate models based upon the Akaike’s Information Criterion corrected for small sample size (AICc). The AICc was used
as it considers both model fit and complexity (number of parameters) to rank models (Burnham & Anderson 2002). For all candidate models, I also calculated the ΔAICc (difference between the AICc of the model of interest and lowest AICc of all candidate models), Akaike weight (\(w_i\); relative support of a particular model based on all candidate models) and the amount of null deviance explained by the model (% dev; Burnham & Anderson 2002; Zuur et al. 2009). All models that performed better than the null model (no covariates included) and showed considerable support (ΔAICc < 4; Burnham & Anderson 2002) were presented for consideration. Based upon model composition and the above criteria I selected the ‘best fit’ model(s) and presented smoothing plots for all included variables. All models were generated and plotted using packages mgcv (Wood & Scheipl 2014), MuMin (Barton 2014) and ggplot2 (Wickham 2009) in the statistical program R (R Core Team 2015). To further aid in the interpretation of model results, I presented the overall change in response and predictor variables over time.

3.3. Results

3.3.1. Vertebrate abundance
Across all survey periods, 10 606 trap nights resulted in 212 and 333 captures of mammals and reptiles, respectively. Seven species of mammals and 24 species of reptiles were identified (Table 3.3). In addition, 16 individual frogs were captured, including Cyclorana maini, Limnodynastes spenceri, Notoden nichollsi and Uperoleia micromeles. All mammals were marked by texta pen to identify within survey recaptures, and individuals were counted only once for each survey period. No individual identification was made for reptiles; however, as a consequence of the low site and survey capture rates in species (maximum
possible site recapture in a survey = 2) it was deemed highly unlikely to affect results.

During the data collection phase of this study the *Eremiascincus* species were reclassified into several new species (Mecke et al. 2013). All seven individuals caught after the reclassifications were identified as *E. intermedius*; however, retrospective classification could not be made definitively on the two individuals caught prior. To ensure data accuracy all individuals were analysed together as *Eremiascincus* spp., although based on photographic evidence it is likely all individuals were *E. intermedius*. 
Table 3.3 Mammal and reptile captures during all surveys between November 2012 and November 2014. Detection rate is the proportion of times the species was detected across all sites and survey periods. Bold indicates species with sufficient data for individual abundance modelling.

<table>
<thead>
<tr>
<th>Class</th>
<th>Species</th>
<th>Family</th>
<th>Number captures</th>
<th>Detection rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mammal</td>
<td><em>Pseudomys desertor</em></td>
<td>Muridae</td>
<td>31</td>
<td>0.16</td>
</tr>
<tr>
<td></td>
<td><em>Mus musculus</em></td>
<td>Muridae</td>
<td>10</td>
<td>0.06</td>
</tr>
<tr>
<td></td>
<td><em>Sminthopsis youngsoni</em></td>
<td>Dasyuridae</td>
<td>5</td>
<td>0.05</td>
</tr>
<tr>
<td></td>
<td><em>Dasycercus blythi</em></td>
<td>Dasyuridae</td>
<td>2</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td><em>Pseudomys hermannsburgensis</em></td>
<td>Muridae</td>
<td>101</td>
<td>0.33</td>
</tr>
<tr>
<td></td>
<td><em>Notomys alexis</em></td>
<td>Muridae</td>
<td>45</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td><em>Sminthopsis macroura</em></td>
<td>Dasyuridae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td>Reptile</td>
<td><em>Rhynchoedura ornata</em></td>
<td>Gekkonidae</td>
<td>5</td>
<td>0.05</td>
</tr>
<tr>
<td></td>
<td><em>Carlia triacantha</em></td>
<td>Scincidae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td><em>Ctenophorus isolepis</em></td>
<td>Agamidae</td>
<td>2</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td><em>Ctenotus grandis</em></td>
<td>Scincidae</td>
<td>5</td>
<td>0.05</td>
</tr>
<tr>
<td></td>
<td><em>C. hanloni</em></td>
<td>Scincidae</td>
<td>57</td>
<td>0.30</td>
</tr>
<tr>
<td></td>
<td><em>C. inornatus</em></td>
<td>Scincidae</td>
<td>5</td>
<td>0.03</td>
</tr>
<tr>
<td></td>
<td><em>C. pantherinus</em></td>
<td>Scincidae</td>
<td>12</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td><em>C. schomburgkii</em></td>
<td>Scincidae</td>
<td>60</td>
<td>0.34</td>
</tr>
<tr>
<td></td>
<td><em>Delma tincta</em></td>
<td>Pygopodidae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td><em>Diplodactylus conspicillatus</em></td>
<td>Gekkonidae</td>
<td>2</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td><em>Liopholis striata</em></td>
<td>Scincidae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td><em>Eremiascincus spp.</em></td>
<td>Scincidae</td>
<td>9</td>
<td>0.06</td>
</tr>
<tr>
<td></td>
<td><em>Heteronotia binoei</em></td>
<td>Gekkonidae</td>
<td>5</td>
<td>0.04</td>
</tr>
<tr>
<td></td>
<td><em>Strophurus elderi</em></td>
<td>Gekkonidae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td><em>S. ciliaris</em></td>
<td>Gekkonidae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td><em>S. jeanae</em></td>
<td>Gekkonidae</td>
<td>5</td>
<td>0.04</td>
</tr>
<tr>
<td></td>
<td><em>Lerista bipes</em></td>
<td>Scincidae</td>
<td>81</td>
<td>0.31</td>
</tr>
<tr>
<td></td>
<td><em>L. labialis</em></td>
<td>Scincidae</td>
<td>34</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td><em>Menetia greyii</em></td>
<td>Scincidae</td>
<td>3</td>
<td>0.03</td>
</tr>
<tr>
<td></td>
<td><em>Notoscincactus ornatus</em></td>
<td>Scincidae</td>
<td>26</td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td><em>Anilios endoterus</em></td>
<td>Typhlopidae</td>
<td>2</td>
<td>0.02</td>
</tr>
<tr>
<td></td>
<td><em>Pseudonaja modesta</em></td>
<td>Elapidae</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td><em>Lucasium stenodactylum</em></td>
<td>Gekkonidae</td>
<td>13</td>
<td>0.08</td>
</tr>
<tr>
<td></td>
<td><em>Varanus gouldi</em></td>
<td>Varanidae</td>
<td>1</td>
<td>0.01</td>
</tr>
</tbody>
</table>
Examination of the relationships between species indicated five significant positive relationships between the species with sufficient captures (Table 3.4). However, the overall strength of the relationships were weak (coefficients < 0.4; Evans 1996) with the exception of the moderately strong relationship between the two *Lerista* species.

Table 3.4 Spearman's rank-order correlations between vertebrate ordinal abundance (# individuals) across all surveys. Bold indicates significance; * P < 0.05, ** P < 0.01, *** P < 0.001.

<table>
<thead>
<tr>
<th></th>
<th>P. desertor</th>
<th>P. herman.</th>
<th>C. hanloni</th>
<th>C. schomb.</th>
<th>N. ornatus</th>
<th>L. bipes</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>P. herman.</em></td>
<td>0.13</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C. hanloni</td>
<td>-0.05</td>
<td>0.08</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C. schomb.</td>
<td>0.04</td>
<td>0.03</td>
<td>0.24*</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>N. ornatus</td>
<td>0.02</td>
<td>0.05</td>
<td>0.19</td>
<td>0.18</td>
<td></td>
<td></td>
</tr>
<tr>
<td>L. bipes</td>
<td>-0.17</td>
<td>0.01</td>
<td>0.25</td>
<td>0.34**</td>
<td>0.32*</td>
<td></td>
</tr>
<tr>
<td>L. labialis</td>
<td>0.03</td>
<td>0.20</td>
<td>0.24</td>
<td>0.30*</td>
<td>0.19</td>
<td>0.42***</td>
</tr>
</tbody>
</table>

3.3.2. *Habitat structure, fire, invertebrates and climate statistics*

PCA analysis on the habitat structure data resulted in two components explaining 88.59% of variation between sites and surveys. In the two components only three variables; *Triodia*, bare ground and shrubs, strongly contributed to the differences between sites and surveys (loadings > 0.3).

Unsurprisingly, the level of *Triodia* cover showed strong correlations with fire variables and the amount of bare ground. Due to these correlations, bare ground and fire were removed from all final models. *Triodia* cover on burnt sites was low at the beginning of sampling (~15%) and increased steadily over time (to ~25%) whilst partially burnt consistently recorded an average of 30% cover across the entire study. At unburnt sites *Triodia* cover was high initially (~45%) and reduced slightly over time (to ~40%).
Both the hymenoptera index and other invertebrate index did not change across survey periods; however, large variations were found between individual sites. This variation was not the result of any consistent differences at particular sites but rather it resulted from peaks of different orders at different sites during different survey periods.

Rainfall, temperature and large mammalian predator presence varied considerably over time. The two large predators; cats and wild dogs, showed an inverse relationship until February 2014 after which both increased steadily until the completion of the study (Figure 3.2). Overall, cat presence appeared to show annual peaks in November whilst wild dog presence appeared to increase in May each year. Larger than normal monthly rainfall (over 100 mm) was recorded in May 2013 and January 2014. Temperature varied as expected with low monthly temperatures in the winter months (May and Aug) and high monthly temperatures recorded in the summer months (Nov and Feb) (Figure 3.3).

Figure 3.2 Percentage of sites across Newhaven Wildlife Sanctuary with predator sign detected over all survey periods. Solid line (cats); dotted line (wild dog).
3.3.3. Mammal response

The variation in species richness of small mammals was due to variations in both dasyurid and rodent diversity. However, across the majority of surveys a greater diversity of rodent species were caught (Figure 3.4a). The two rodent species individually examined here, *Pseudomys desertor* and *P. hermansburgensis*, showed relatively different patterns of abundance over time (Figure 3.4b). After the initial two survey periods *P. hermansburgensis* decreased to very low numbers and then increased dramatically in the last two survey periods. *P. desertor* showed a steady decrease across the entire period surveyed, with a minimal increase in capture rates in the last two surveys.
Figure 3.4 Change over all survey periods of (a) species richness of Dasyuridae (purple) and Rodentia (green) and (b) total captures of *P. desertor* (Red) and *P. hermannsburgensis* (Blue) across all sites.

Only a single model, which included predator and short-term climate, was found to have considerable support for predicting mammal richness, whilst for *Pseudomys desertor* and *P. hermannsburgensis* five and two models were found, respectively (Table 3.5; Appendix 2). The first ranked models for both *P. desertor* (vegetation) and *P. hermannsburgensis* (predators and long-term climate) were deemed the ‘best fit’ models as all other models showed considerably lower
Akaike weights and did not substantially increase the degree of deviance explained by the model.

Table 3.5 Model selection summary statistics from small mammal GAMMS with considerable support (ΔAICc < 4). Model covariates indicate variable groups as specified in Table 3.2. df = degrees of freedom in model, AICc = Akaike’s Information Criterion corrected for small sample size, ΔAICc = difference from lowest AICc in candidate models, wi = Akaike weight of model and % Dev = percentage null deviance explained by model. The ‘best fit’ model as defined are underlined.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>Rank – Model covariates</th>
<th>df</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>wi</th>
<th>% Dev</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Mammal richness</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Predator + Short term climate</td>
<td>22</td>
<td>223.07</td>
<td>0.00</td>
<td>0.94</td>
<td>76.42</td>
<td></td>
</tr>
<tr>
<td><strong>Pseudomys desertor</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Vegetation</td>
<td>11</td>
<td>117.57</td>
<td>0.00</td>
<td>0.45</td>
<td>68.41</td>
<td></td>
</tr>
<tr>
<td>2 – Vegetation + Interaction</td>
<td>13</td>
<td>119.26</td>
<td>1.69</td>
<td>0.19</td>
<td>70.39</td>
<td></td>
</tr>
<tr>
<td>3 – Vegetation + Short term climate</td>
<td>14</td>
<td>120.24</td>
<td>2.67</td>
<td>0.12</td>
<td>72.41</td>
<td></td>
</tr>
<tr>
<td>4 – Vegetation + Long term climate</td>
<td>14</td>
<td>120.69</td>
<td>3.11</td>
<td>0.10</td>
<td>71.08</td>
<td></td>
</tr>
<tr>
<td>5 – Vegetation + Invertebrates</td>
<td>14</td>
<td>121.04</td>
<td>3.47</td>
<td>0.08</td>
<td>70.56</td>
<td></td>
</tr>
<tr>
<td><strong>Pseudomys hermannsburgensis</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Predators + Long term climate</td>
<td>19</td>
<td>234.27</td>
<td>0.00</td>
<td>0.70</td>
<td>59.52</td>
<td></td>
</tr>
<tr>
<td>2 – Predators + Short term climate</td>
<td>19</td>
<td>237.08</td>
<td>2.81</td>
<td>0.17</td>
<td>57.56</td>
<td></td>
</tr>
</tbody>
</table>

Across all small mammal GAMMs the variable groups’ invertebrates and interactions were not included in any of the models considered ‘best fit’ (Figure 3.5). The proportion of *Triodia* cover was the only variable deemed a good predictor of *P. desertor* abundance (edf = 3.01, p = 0.04) with lower abundance when *Triodia* cover was below 40%. Large predators were an important predictor for both small mammal richness (wild dog; edf = 1.00, p < 0.001) and *P. hermannsburgensis* (cat: edf = 2.42, p < 0.001; wild dog: edf = 1.51, p < 0.001). A consistent positive relationship was found between the proportion of sites wild
dogs were present and both mammal richness and *P. hermannsburgensis* abundance. The response of *P. hermannsburgensis* to cats was considerably different. After an initial positive response, *P. hermannsburgensis* abundance peaked when cat presence reached 40% of sites and then declined once cat presence was detected at more than 50% of sites.

Temperature variables were found to be significant for both mammal richness (edf = 3.08, p < 0.001) and *P. hermannsburgensis* (edf = 1.70, p < 0.001; Figure 3.6). Small mammal richness was lowest when minimum temperatures within 4 days of sampling was approximately 15 °C but increased when above and below this point. Increases in the maximum temperatures 11 months prior to sampling were a consistent positive predictor of *P. hermannsburgensis* abundance.
Figure 3.5 Smoothing plots for non-climate variables present in top GAMMS for small mammal. Solid line = GAMM model fit; dotted line indicates 95% confidence bands; plots of significant variables (p > 0.05) are shown in bold. Missing plots indicate variables not present in top models.
Figure 3.6 Smoothing plots for climate variables present in top GAMMS for small mammals. Solid line = GAMM model fit; dotted line indicates 95% confidence bands; plots of significant variables (p > 0.05) are shown in bold. Missing plots indicate variables not present in top models.
3.3.4. Reptile response
The diversity of reptiles caught was influenced primarily by changes in diversity of the families Scincidae and Geckonidae (Figure 3.7a). All reptile species showed seasonal fluctuations in captures rates. Capture rates were low in the May and August surveys and high in November and February (Figure 3.7b).

Figure 3.7 Change over all survey periods for (a) species richness of Agamidae (orange), Elapidae (brown), Geckonidae (dark blue), Pygopodidae (purple), Scincidae (green), Typhlopidae (grey) and Varanidae (black) and (b) abundance of C. hanloni (Red), C. schomburgkii (Army green), L. bipes (Aqua), L. labialis (Blue) and N. ornatus (Pink) across all surveys.
As predicted, temperature during surveys was a strong positive predictor for reptile richness and all individual species abundances (Appendix 3). Only a single model was found to have considerable support for reptile richness (predator and long-term climate), *Notoscincus ornatus* (predator and invertebrates) and *Lerista labialis* (predator and invertebrates; Table 3.6). Two models were selected as equal ‘best fit’ for *Ctenotus hanloni* (both with predators and climate), as both showed similar Akaike weights and explained similar levels of null deviance. A single ‘best fit’ model was selected for *L. bipes* (predators) as all other models showed considerably lower Akaike weights and did not substantially increase the degree of deviance explained by the model. None of the candidate models performed better than the null model for predicting *C. schomburgkii* abundance.
Table 3.6 Model selection summary statistics of reptile GAMMS with considerable support (ΔAICc < 4). df = degrees of freedom in model, AICc = Akaike’s Information Criterion corrected for small sample size, ΔAICc = difference from lowest AICc in candidate models, wi = Akaike weight of model and % Dev = percentage null deviance explained by model. The ‘best fit’ model as defined are underlined.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>Rank – Model covariates</th>
<th>df</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>wi</th>
<th>% Dev</th>
</tr>
</thead>
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<tr>
<td><strong>Reptile richness</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Predator + Long term climate</td>
<td>18</td>
<td>306.35</td>
<td>0.00</td>
<td>0.88</td>
<td>68.18</td>
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</tr>
<tr>
<td><strong>Ctenotus hanloni</strong></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Predator + Short term climate</td>
<td>15</td>
<td>166.89</td>
<td>0.00</td>
<td>0.39</td>
<td>68.13</td>
<td></td>
</tr>
<tr>
<td>2 – Predator + Long term climate</td>
<td>17</td>
<td>167.95</td>
<td>1.06</td>
<td>0.23</td>
<td>70.08</td>
<td></td>
</tr>
<tr>
<td>3 – Vegetation + Short term climate</td>
<td>7</td>
<td>169.27</td>
<td>2.37</td>
<td>0.12</td>
<td>51.33</td>
<td></td>
</tr>
<tr>
<td>4 – Vegetation + Predator</td>
<td>13</td>
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<td>2.75</td>
<td>0.10</td>
<td>60.34</td>
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<td><strong>Ctenotus schomburgkii</strong></td>
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<td>1 – Null model</td>
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<td><strong>Notoscinctus ornatus</strong></td>
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<tr>
<td>1 – Predator + Invertebrates</td>
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<td>0.85</td>
<td>98.55</td>
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<tr>
<td><strong>Lerista bipes</strong></td>
<td></td>
<td></td>
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<tr>
<td>1 – Predators</td>
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<td>0.53</td>
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<tr>
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<td>2.18</td>
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</tr>
<tr>
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<tr>
<td>4 – Predators + Competitors</td>
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<td>0.08</td>
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<tr>
<td><strong>Lerista labialis</strong></td>
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</tr>
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<td>1 – Predators + Invertebrates</td>
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<td>0.00</td>
<td>0.99</td>
<td>83.97</td>
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</tbody>
</table>

Across all reptile GAMMs the variable groups’ vegetation and interaction were not included in any of the models deemed ‘best fit’. Predator presence was the most consistent predictor for reptile richness and individual species abundance (Figure 3.8). Increasing cat presence was found to have a consistent positive association with reptile richness (edf = 1.00, p < 0.001), *C. hanloni* abundance (edf = 1.00, p = 0.001) and *L. bipes* abundance (edf = 1.00, p < 0.001). Wild dog
presence was considered an important predictor for reptile richness and all individual species abundance except *C. schomburgkii*. Reptile richness (edf = 1.00, p = 0.002) and the abundances of *N. ornatus* (edf = 1.79, p = 0.001) and *L. bipes* (edf = 1.00, p < 0.001) decreased steadily as wild dog presence increased over sites. *C. hanloni* abundance also decreased once wild dog presence was recorded at between 45 % and 55 % of sites (top ranked model; edf = 2.38, p < 0.001, second ranked model; edf = 2.78, p = 0.007). In contrast, *L. labialis* abundance was lowest when wild dog presence was recorded at 50 - 60% of sites but increased when above and below these levels.

Hymenoptera and other invertebrate indices were found to be a good predictors of *L. labialis* abundance (Hymenoptera; edf = 2.91, p = 0.002, other invertebrate; edf = 2.86, p = 0.007). The abundance of *L. labialis* decreased once the Hymenoptera index was greater than 2000 and increased when the other invertebrate index was recorded above 1300. Notably, the responses to both invertebrate variables varied considerably between sites and surveys when a larger invertebrate index was recorded.

Rainfall totals were significant variables in predicting reptile richness and *Ctenotus hanloni* (Figure 3.9). The top ranked model for *C. hanloni* showed a steady increase in abundance when greater rainfall was recorded within 14 days of sampling (edf = 1.00, p < 0.001). The second ranked model also showed a positive response to rainfall with greater *C. hanloni* abundance being recorded when the average rainfall event increased to 40 mm in the previous 11 months (edf = 2.42, p = 0.02). Overall, reptile richness also increased as the number of rain days recorded the previous month increased to seven (edf = 2.41, p < 0.001) and stabilised when the number of rain days exceeded this.
Figure 3.8 Smoothing plots for non-climate variables present in top GAMMS for reptiles. Solid line = GAMM model fit; dotted line indicates 95% confidence bands; plots of significant variables (p > 0.05) are shown in bold. Missing plots indicate variables not present in top models. Black indicates model with rank 1; Grey indicates models ranked 2.
Figure 3.9 Smoothing plots for climate variables present in top GAMMS for reptiles. Solid line = GAMM model fit; dotted line indicates 95% confidence bands; plots of significant variables (p > 0.05) are shown in bold. Missing plots indicate variables not present in top models. Black indicates top ranked model; Grey indicates second ranked model.
3.4. Discussion

Although capture rates were relatively low for both mammals and reptiles (2% and 3% trap success, respectively), they were comparable to previous studies conducted during periods of low rainfall in spinifex sand plains (e.g. Letnic et al. 2004; Letnic et al. 2005). Based upon long term monitoring in the area (AWC 2016a) most of the expected species were represented in trapping results. However, most species were only caught in very low numbers, particularly dasyurid marsupials, which reflects similar trends seen in the long term monitoring conducted across the property (AWC 2016a). My results strongly supported predictions regarding temperature for both small mammal and reptiles. Strong relationships were also found with predators, although trends were not always as expected. Mammal results supported the initial prediction regarding vegetation whilst reptiles partially supported the predictions relating to rainfall and invertebrates. Notably, none of the variables tested here were considered good predictors of *Ctenotus schomburgkii* abundance. The results are discussed in more detail below.

3.4.1. Response to vegetation variables

The results showed support for predictions regarding the influence of vegetation in relation to small mammals but not reptiles. In agreement with previous research that has shown strong habitat preferences by *Pseudomys desertor*, the degree of *Triodia* cover was a strong positive predictor for *P. desertor* abundance in my study (Kutt et al. 2004; Letnic et al. 2004; Masters 1993; McDonald et al. 2015b; Read et al. 1999). The level of cover at which *P. desertor* abundance was highest (> 40%) was representative of the unburnt areas only, indicating that removal of vegetation by fire, regardless of extent/patchiness, would reduce the abundance of *P. desertor* in the area. In contrast, *P. hermannsburgensis* did not
respond to changes in vegetation cover/fire which is consistent with other studies in similar habitats (i.e. Greenville et al. 2016; Pastro et al. 2011). During drought *P. hermannsburgensis* is considered more mobile (Dickman et al. 2010) and shows larger versatility in diet (Murray & Dickman 1994) and habitat selection (Dickman et al. 2011). This likely enables the species to utilise a broader range of habitats with varying levels of cover. Similarly, none of the reptiles responded to changes in either *Triodia* or shrub cover. Previous studies have shown certain reptiles (particularly *Ctenotus* species) use areas with greater cover in order to facilitate thermoregulation and avoid extremely high temperatures (Letnic et al. 2004; Masters 1996; Pianka & Goodyear 2012). This is likely not as important for the nocturnal and crepuscular species predominantly sampled in this study. Additionally, prior to the beginning of our study burnt sites had already regenerated to low levels of cover (~15%) and therefore may have already provided sufficient levels of cover, as seen in studies conducted on similar species (Masters 1996; Pastro et al. 2011).

### 3.4.2. Response to food resources

Predictions regarding the influence of invertebrates on species with an insectivorous diet showed limited support. Due to low capture rates, I was unable to examine the response of any insectivorous small mammals and only one insectivorous reptile, *L. labialis*, was found to have a strong significant relationship. The lack of a response from many insectivorous reptile species may be a result of a lack of specialisation in prey species. The large fluctuations seen in the invertebrate index were predominantly a consequence of increases in Isoptera, Coleoptera, Isopoda and Aranae. *L. labialis* is known to feed predominantly on termites (Isoptera; Greenville & Dickman 2005; Letnic et al. 2004) and therefore may have been more responsive than other reptile species.
that consume a wider variety of prey, such as *Ctenotus* species (Goodyear & Pianka 2011; Twigg et al. 1996). In addition, the ability of reptiles to respond immediately to increased resources may be limited due to the larger period of time needed to immigrate into areas or respond reproductively, as seen in longer term climate studies (Cadby et al. 2010).

### 3.4.3. Response to climate variables

I found some support for my predictions that physiological requirements would drive the relationship between temperature and small mammals. Small mammal diversity was lowest when temperatures reached around 15 °C, which coincides with the period transitioning between summer and winter. Changes in detection and/or abundance during these periods are likely a reflection of seasonal differences in movement caused by breeding requirements (Krebs et al. 1994), age structure (Krebs et al. 1994; O'Farrell et al. 1994; Tanaka & Kanamori 1969), or overall seasonal fluctuations in population numbers (Masters 1993). *P. hermannsburgensis* also responded positively to increases in long-term temperature (maximum temperature 11 months prior) which may reflect optimal conditions during the previous breeding season. *P. hermannsburgensis* is considered a seasonal and opportunistic breeder (Breed 1990) that responds readily to rainfall (Predavec 1994). During periods of lower rainfall, as in this study, it is likely that *P. hermannsburgensis* follow more seasonal breeding than shown during periods of higher rainfall. *P. hermannsburgensis* also showed no threshold at which maximum temperatures became inhibiting or detrimental, which may be due to their known resilience to limited water and dry environments (Kotler et al. 1998). Overall, small mammals showed no response to rainfall in my study even though several large ‘top up’ rainfall events occurred which may indicate a greater impact of
suppressive influences (particularly predation) during these periods (Kotler et al. 1998). Further comparative studies in areas with varying predator pressure during periods of low productivity would help to clarify this.

In contrast, higher reptile richness was strongly associated with a greater number of rain days recorded within a month of sampling. Reptile assemblages in arid zone Australia are considered one of the most diverse in the world (Morton & James 1988) and it is likely that recent rainfall would promote development of different vegetative states (Letnic et al. 2004) and resource abundance (Pastro et al. 2013), which in turn would better support a greater richness of reptiles. As seen in other dry environments (Ryan et al. 2016), rainfall may also provide essential increased levels of ambient moisture that help promote reptile activity. Variations in reptile richness were predominantly caused by greater numbers of skink and gecko species being detected, particularly *Ctenotus* species. *Ctenotus* activity patterns have been shown to vary based on species specific optimal temperature thresholds which reduce inter-specific competition and support greater diversity (Read 1998). This may also be the case with rainfall, although further exploration, particularly regarding species caught at low numbers here, would help clarify this.

The only species analysed individually that showed a strong relationship with rainfall was *Ctenotus hanloni*, which increased in abundance with greater rainfall over both short (14 days prior) and long (11 months prior) time frames. Rainfall is considered a primary driver in invertebrate abundance (Kwok et al. 2016; Langlands et al. 2006) which would likely increase activity of predators, such as *C. hanloni*, over short periods and breeding success over longer periods (Pastro et al. 2011). The larger body size (compared to the other individual species examined here; Wilson & Swan 2013) and the dietary niche partitioning
found in *Ctenotus* species (Goodyear & Pianka 2011) may have a resulted in the specific response of *C. hanloni* found here. Previous studies have shown larger lizards are capable of subduing larger more hard bodied prey (Verwaijen et al. 2002) which were shown to be more responsive to short term rainfall in a related study on invertebrates in the area (See Chapter 4).

The *a priori* assumption of the relationship between reptiles and temperature was supported, with temperature being a significant component of all reptile models.

### 3.4.4. *Response to predator presence*

Although there was an apparent effect of predators on small mammal and reptile diversity and abundance, it was not always in the direction that the relationship was predicted to occur. In contrast to predictions, reptile species tended to show a positive response to cat presence, whilst the response by *P. hermannsburgensis* was negative overall. The results showed *P. hermannsburgensis* are capable of withstanding low levels of cat presence. However, numbers declined sharply once cats were detected at over 50% of the sites. *P. hermannsburgensis* was the most abundant mammal species recorded (101 captures) and has known to be readily consumed by cats (Dickman 1996b; Doherty 2015; Paltridge 2002). Studies have previously shown, during periods of low rainfall, *P. hermannsburgensis* select areas with lower cat abundance and show higher levels of predator awareness in more open habitats (Dickman et al. 2011) indicating predation is potentially of greater concern compared to other species in the area.

Surprisingly, reptile richness, *C. hanloni* and *L. bipes* all showed a positive response to increased cat presence. This unexpected result may be due to a
seasonal response in the abundance of both reptiles and cats. Cat abundances appeared to peak annually in November (likely due to the entry of juveniles into the system; Read et al. 2001) which coincides with the increased activity of reptiles during warmer months (November – February). Interestingly, cats have previously shown a high ability to prey switch depending on availability (Kutt 2011; Newsome et al. 1983; Paltridge 2002; Robley et al. 2004; Spencer et al. 2014); however, no negative impact was detected here. This coincides with previous predator removal experiments which showed limited responses by reptiles to the absence or presence of cats (Moseby et al. 2009b). As this study examined the most commonly detected species, our results may also be biased to those that are naturally more resilient and capable of withstanding increased predation pressure. Furthermore, ‘top up’ rainfalls during our study may have helped to buffer populations against any potential declines caused by greater predation. Continual examination of the response of small mammals and reptiles to predators would increase our knowledge regarding the effect of ‘top up’ rainfalls on interactions long term.

Considering the predominant inverse relationship between cats and wild dogs in the study site the positive response of small mammals to wild dogs was expected. The potential influence of wild dogs as regulators of cats which promotes greater small mammal diversity and abundance through reducing predatory pressure has been shown in several previous studies (Cremona et al. 2014; Fleming et al. 2012; Letnic et al. 2012). The lack of negative effect is also unsurprising as the occurrence of small mammals in wild dog scats is generally low in drought times and during periods of high rainfall is predominantly due to consumption of feral house mouse (Mus musculus), hopping mice (Notomys spp.) and irruptive species (Allen & Leung 2012; Paltridge 2002). None of these
preferred species or groups was detected in high numbers in this study and therefore it is not surprising that wild dogs had little negative effect on small mammals populations sampled here.

A consistent negative effect was found between wild dog presence and reptile richness and individual species abundances. Although previous studies have shown dingoes (referred to as wild dogs here) in the region readily consume skinks, it has not been possible to identify prey past family level (Paltridge 2002) making it difficult to ascertain the degree of predator pressure imposed on individual species. Furthermore, the negative relationship found here might be due to communal seasonal fluctuations. Wild dog activity was greatest in May (likely due to the changes in behaviour during breeding; Catling et al. 1992) which coincides with the regional reduction in reptile activity during cooler months (May – September). Further investigation into the potential negative impact of wild dogs on reptiles is warranted to determine whether the results here are a genuine reflection of interactions in the system. Interestingly, wild dog presence had a different effect on *Lerista labialis* abundance, which decreased until wild dogs were recorded at 60% of sites, and increased when wild dog presence exceeded this. Due to the small size and subterranean lifestyle of *L. labialis* (Greenville & Dickman 2005) it is unlikely to be of particular importance in wild dog diet and therefore this relationship is likely due to the communal response to a variable not tested here.

### 3.4.5. Conclusions

Overall, this study provided important insights into the relationships between vertebrate species and environmental and climatic variables that occur during a post boom period in spinifex sand plains of arid Australia. Much of the previous literature on Australian small mammal and reptile responses to rainfall in arid
regions has focused on areas of higher latitudes that experience, on average, lower levels of rainfall (e.g. Bennison et al. 2013; Greenville & Dickman 2005; Greenville et al. 2012; Letnic & Dickman 2005; Masters 1993; Pavey & Nano 2013b). The ‘top up’ rainfalls experienced at my site might have maintained ecosystem functioning at a higher level and therefore buffered against some of the dramatic declines noted in other studies (Greenville et al. 2012; Letnic & Dickman 2006; Letnic et al. 2005; Letnic et al. 2013). This indicates an important potential difference between sites in southern arid Australia and locations under a more monsoonal influence such as Newhaven. During periods of low rainfall, cats and wild dogs were consistently considered important factors influencing the persistence of both small mammal and reptiles, respectively. Although concerning, I was unable to assess predator activity/abundance in detail due to the time limitations during field surveys. More in-depth analysis of predator activity and abundance in relation to small mammal and reptile persistence in a post boom period, particularly in regions where rainfall is higher, would help to clarify the strength of the interactions found here.
CHAPTER 4

SPECIES OCCURENCE IN A POST BOOM PERIOD
PART B: INVERTEBRATES

Clockwise from top left: Golden orb weaving spider (J Molyneux), Leopard grasshopper (J Molyneux), Desert scorpion (A Molyneux), Microscopic view of Assassin bug (J Molyneux), Blistered pyromorph (J Molyneux).
Chapter 4 | Invertebrate occurrence

Abstract
Current knowledge on the biology, assemblage and drivers of Australia’s arid zone invertebrates is sparse. They are largely forgotten in management strategies even though they play many key roles in ecosystem functioning. From studies conducted in arid zones across the world and the limited number of studies from Australia, the primary drivers of invertebrate occurrence are thought to be climate, particularly temperature and rainfall. However, the influence of these drivers varies with vegetation and interactions between and within taxa. This study investigated the effect of multiple potential drivers of invertebrate orders in the spinifex sand plains of central Australia. The study was conducted in a post boom period following the dramatic decline of most vertebrate species in the area. Results supported the trends found globally. Rainfall and temperature were the most consistent drivers for the orders examined here. Vegetation cover was not a good predictor of any invertebrate species; however, increased abundance of large vertebrate predators (native and non-native) coincided with greater abundance of many invertebrate orders tested here. Areas that support greater invertebrate abundance likely attract more predators, particularly during periods when rainfall and resources are decreasing, and potentially play an important role in sustaining their populations during these periods. Interestingly, no suppressive effects were found from any large vertebrate predators, indicating many invertebrates are capable of withstanding the higher levels of predation experienced during low rainfall periods. Overall, this study sampled a large diversity of invertebrates. It highlights the important roles they play in maintaining functioning ecosystems and the need to gain a greater understanding of this little known group.
4.1. **Introduction**

Our knowledge of Australia’s arid zone fauna has predominantly focused on vertebrates species during periods of high rainfall. There is a growing understanding of their ecology during the low phase of population cycles (e.g. Dickman et al. 2011; Pavey et al. 2014). In comparison, limited information is available on the occurrence, ecology and assemblage structure of arid invertebrates. This is a serious shortcoming given the ecological importance of invertebrates in ecosystems (Prather et al. 2013).

Although recognized as important components of arid zone systems (e.g. Polis 1991), relatively little is known regarding the drivers of invertebrate abundance and distribution. Invertebrates play many critical roles and have key influences on both primary production and overall ecosystem functioning (Polis 1991; Prather et al. 2013). Although largely forgotten in management strategies, understanding the processes that drive and support invertebrate populations is essential in order to support sustainable ecosystems into the future. In general, temperature and rainfall are considered primary drivers of invertebrates in arid zones globally (Langlands et al. 2006; Palmer 2010); however, varying responses to changes in habitat structure and soil composition have also been found (Barrow et al. 2007; Gollan et al. 2009; Kwok & Eldridge 2015; Langlands et al. 2006; Langlands et al. 2012; Smith & Morton 1990; Teasdale et al. 2013). Intra and inter specific predation is also a key component in invertebrate dynamics, due to the high number of invertebrate predators (such as spiders, scorpions and centipedes; Kwok & Eldridge 2015; Langlands et al. 2011; Langlands et al. 2006) and vertebrate predators (Gibson 2001; Gilfillan 2001; Haythornthwaite 2005; Pavey et al. 2012; Pavey et al. 2016) playing regulatory roles.
While recognising complex relationships exist, most ecological studies on invertebrates have examined responses to single processes in the environment, such as fire or rainfall, although there are exceptions (e.g. Langlands et al. 2006). A few studies have compared multiple processes simultaneously to determine the most important drivers across several factors. Recently, Kwok et al. (2016) examined the effect of rainfall, temperature, topography and vegetation in a long term (6 year) study in the Simpson Desert. This study showed vegetation to be the most consistent predictor of invertebrate abundance with a comparatively limited response to rainfall. My study involves a similar analysis to Kwok et al. (2016). I examined the factors influencing invertebrate abundance in the sand plains of the southern Tanami, central Australia. The study is focussed on the period of high ecological stress in vertebrates (see Chapter 3); the period immediately post-boom. By examining the relationships within the ecosystem at this time, I aim to determine the factors that influence invertebrate occurrence when resource availability is at its lowest and stressors such as fire and predators are at their peak.

First, I examined potential inter-ordinal relationships to determine general patterns of invertebrate occurrence. Second, I examined a variety of factors that could potentially influence the occurrence of individual orders as well as overall invertebrate richness. I simultaneously assessed the impact of several competing factors including rainfall, temperature, vegetation composition, fire and predation. The rationale for selecting these components is outlined in Chapter 3. I predict that:

1. Short term temperature will play a key role in the occurrence of all invertebrate orders, whilst orders known to prefer humid/wet environments, such as Isoptera, Isopoda, Thysanura and Chilopoda
(CSIRO Division of Entomology 1991; Marshall Cavendish Corporation [MCC] 2003), will be more likely to show strong responses to short term rainfall.

2. Invertebrates will initially respond positively to increases in temperature and rainfall; however, will reach thresholds; above which continual increases will have limited or a negative effect.

3. Non-predatory orders that require vegetation for food and/or reproduction, such as Orthoptera, Blattodea and Coleoptera (Kwok et al. 2016; Teasdale et al. 2013); will respond more strongly to a reduction in vegetation and increased fire.

4. Predator pressure from smaller (native) vertebrates will be limited as overall environmental conditions will reduce their abundance whilst larger mammalian predators will have a greater negative effects on larger invertebrates, such as Orthoptera and Coleoptera, as preferred prey items (small mammals and reptiles) become scarcer (Paltridge 2002).

5. Predatory orders such as Aranae and Chilopoda will respond differently to environmental changes to maximize niche partitioning of resources whilst responding positively to smaller prey orders (e.g. Thysanura).

4.2. Methods

4.2.1. Study site

Newhaven Wildlife Sanctuary is 363 km north-west of Alice Springs, located on the eastern boundary of the Great Sandy Desert Bioregion. The area is considered semi-arid with a mean rainfall of around 300 mm.
The landscape is predominantly spinifex sand plains habitat, which was the focal habitat of this study. The sand plains are dominated by *Triodia pungens*, with intermittent, low density pockets of low shrubs. Common shrubs in the area include *Melaleuca glomerata* and *Grevillea juncifolia*. Forbs and other small grasses also become prevalent immediately following rainfall in recently burnt areas (See Chapter 2 for more detailed description of study site).

This study was conducted between November 2012 and November 2014. Annual rainfall increased from 187.6 mm in 2012 to 381.4 mm in 2014 (BOM 2015b). During the survey period higher than expected rains occurred during May 2013 (122 mm) and January 2014 (189.5 mm).

4.2.2. *Invertebrate sampling*

Twelve sites were selected in the spinifex sand plains habitat at Newhaven (See details in Chapter 3). Sites were a minimum of 1.5 km from each other to ensure spatial independence of sampling in each survey period.

Each site was surveyed quarterly for invertebrates from November 2012 until November 2014. Only ground dwelling nocturnal invertebrates were sampled due to logistical constraints and to coincide with a related study on the diet of small mammals and reptiles (See Chapter 3). I used a combination of small and large pitfalls during each survey period to maximize capture rates across orders. At each site, four large pitfalls (16 cm diameter, 60 cm deep) and drift fences were deployed (See Chapter 3 for details). In addition, at each site four small pitfalls were deployed consisting of plastic containers (75 mm diameter, 130 mm deep) buried flush to the ground. The four small pitfall traps were positioned at random locations in each trapping grid but not within 1 metre of any other trap (including large pitfalls and box traps; See Chapter 3). As a consequence of the
large number of small skinks in the region (See Chapter 2) and their sensitivity to preservative agents, dry pitfalls were used. To prevent animals from escaping I coated to upper inside wall of all small pitfalls with Fluon PTFE (Livefoods unlimited, Tinbeerwah, QLD; Graham & Poland 2012). Between trapping surveys pitfalls were closed with lids and covered with sand and the drift fences from large pitfalls removed. During each survey, all pitfalls were checked early morning, closed during the day and opened late afternoon over three consecutive days. Invertebrates were cleared from traps at the end of each survey period and stored in 70% ethanol until identification. Each invertebrate was identified to order, as taxonomic information from the region is limited and I could not confidently separate individuals past ordinal level. At the completion of each survey period, I calculated each site’s invertebrate ordinal richness (# orders captured) for further analysis. All orders with over 100 captures and those that could be representatively sampled by pitfalls were examined individually for ordinal specific responses.

4.2.3. Fire, Habitat structure, Vertebrate predator and Climate data
As I was interested in the effect of fire on invertebrate populations, site locations were selected based upon recent fire history (See Chapter 3 for details of site selection). Data collection methods regarding habitat structure, large mammalian predators and climate were identical to those described in Chapter 3. In addition to large mammalian predators (cats and wild dogs), I also included a measure of smaller native predators by including the abundance of small mammals and reptiles caught at each site (for sampling protocols of small mammals and reptiles see Chapter 3).
4.2.4. *Statistical analysis*

To investigate the potential correlations of abundances between related orders I calculated Spearman’s rank-order correlation coefficients of all pairwise combinations of ordinal abundance. I then examined each combination based upon the significance (p < 0.05) and strength of the coefficient (< 0.4 = weak relationship, 0.4 – 0.6 = moderate relationship, > 0.6 strong relationship; Evans 1996). I used Spearman’s rank-order correlation as it allowed for non-normal distributions in abundances. I investigated the influence of the various environmental variables on invertebrate ordinal abundance and richness through generalised additive mixed models (GAMMs). I utilised GAMMs as it allowed for the repeated measures trapping design used and because initial data exploration showed clear non-linear relationships between response and predictor variables (Zuur et al. 2014). To reduce the number of variables in models I conducted principal component analysis (PCA) on all vegetation and ground cover variables to determine which contributed to the most site-specific differences. As cumulative climate variables showed large collinearity (Spearman rank-order correlations > 0.60), I used only the variable with the strongest correlation to the response variable in each final model (Table 4.1). This resulted in four climate variables being included in each model; short term temperature, long-term temperature, short-term rainfall and long-term rainfall. All continuous covariates were standardized to account for the different scales of measurement between covariates (Zuur et al. 2009).
Table 4.1 Climate variables used in generalised additive mixed models for each response variable examined. Numbers in parentheses indicate number of days (short term) or months (long term) included in accumulated totals.

<table>
<thead>
<tr>
<th>Model</th>
<th>Temperature</th>
<th>Rainfall</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Short term</td>
<td>Long term</td>
</tr>
<tr>
<td>Richness</td>
<td>Min. (13)</td>
<td>Min. (12)</td>
</tr>
<tr>
<td>Aranae</td>
<td>Min. (17)</td>
<td>Min. (1)</td>
</tr>
<tr>
<td>Blattodea</td>
<td>Min. (13)</td>
<td>Min. (1)</td>
</tr>
<tr>
<td>Chilopoda</td>
<td>Min. (14)</td>
<td>Min. (1)</td>
</tr>
<tr>
<td>Coleoptera</td>
<td>Min. (14)</td>
<td>Min. (15)</td>
</tr>
<tr>
<td>Ants</td>
<td>Min. (11)</td>
<td>Max. (13)</td>
</tr>
<tr>
<td>Isopoda</td>
<td>Min. (1)</td>
<td>Max. (13)</td>
</tr>
<tr>
<td>Orthoptera</td>
<td>Min. (13)</td>
<td>Max. (13)</td>
</tr>
<tr>
<td>Thysanura</td>
<td>Max. (14)</td>
<td>Max. (2)</td>
</tr>
</tbody>
</table>

Due to the limitations of the data and to avoid overfitting, I was unable to include all variables simultaneously. Therefore variables were grouped by ecological relevance (Table 4.2) to create a set of 15 candidate models that comprised all single and pair-wise combinations of the variable groups (including the null model; Appendix 4).

Table 4.2 Grouping of variables for candidate model structure based upon ecological relationship to response variables.

<table>
<thead>
<tr>
<th>Group</th>
<th>Variables included in group</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation</td>
<td>All vegetation variables</td>
</tr>
<tr>
<td>Lrg predators</td>
<td>Cat presence</td>
</tr>
<tr>
<td></td>
<td>Wild dog presence</td>
</tr>
<tr>
<td>Sml predators</td>
<td>Small mammal abundance</td>
</tr>
<tr>
<td></td>
<td>Reptile abundance</td>
</tr>
<tr>
<td>Short term climate</td>
<td>Short term temperature</td>
</tr>
<tr>
<td></td>
<td>Short term rainfall</td>
</tr>
<tr>
<td>Long term climate</td>
<td>Long term temperature</td>
</tr>
<tr>
<td></td>
<td>Long term rainfall</td>
</tr>
</tbody>
</table>
All variables were checked for outliers and collinearity using Pearson correlation coefficients (< 0.70) and variance inflation factors (< 3; Zuur et al. 2014). If two variables showed strong correlation, I retained the variable considered most ecologically relevant based upon previous knowledge of the response variable. All models included site as a random factor, to account for repeated sampling. Initially log (number of trap nights) was included as a covariate to account for variations in trap effort between sites and surveys; however, as estimated parameters were close to one I followed the recommendations of Zuur et al. (2014) and included it as an offset in models. The offset function assumes a proportional relationship with the response variable meaning greater trap effort resulted in a higher chance of detection.

GAMM models were initially generated with Poisson distribution and a log link function and tested for over-dispersion (pearson statistics > 1; Zuur et al. 2012). If over-dispersion was apparent, the model was re-run using a negative binomial distribution with a log link function which accounted for the additional variance in the data by the addition of an extra parameter (k or theta; Zuur et al. 2009).

All candidate models were ranked based upon the Akaike’s Information Criterion corrected for small sample size (AICc). The AICc was used as it considers both model fit and complexity (number of parameters) to rank models (Burnham & Anderson 2002). For all candidate models I also calculated the ΔAICc (difference between the particular models AICc and lowest AICc of all candidate models), Akaike weight (wi; relative support of a particular model based on the models tested) and the amount of null deviance explained by the model (% dev; Burnham & Anderson 2002; Zuur et al. 2009). All models that performed better than the null model (no covariates included) and showed considerable support (ΔAICc < 4; Burnham & Anderson 2002) were presented.
for consideration. Based upon model composition and the above criteria I selected the ‘best fit’ model(s) and presented smoothing plots for all included variables. All models were generated and plotted using packages; *mgcv* (Wood & Scheipl 2014), *MuMin* (Barton 2014) and *ggplot2* (Wickham 2009) in the statistical program *R* (R Core Team 2015).

4.3. Results

4.3.1. *Invertebrate abundance and inter-ordinal relationships*

Across all survey periods, 2 784 trap nights resulted in 50 669 captures of invertebrates. A total of 25 orders were identified. Ants (family Formicidae, Hymenoptera) were the most dominant group, representing 88.49% of the samples taken (Table 4.3). Based upon the conditions specified above, eight invertebrate orders were analysed individually; one from the class Arachnida: Aranae (spiders); one from the phylum Crustacea: Isopoda (slaters); one from the subphylum Myriapod: Chilopoda (centipedes) and five from the class Insecta: Blattodea (cockroaches), Coleoptera (beetles), Hymenoptera (ants), Orthoptera (crickets and grasshoppers) and Thysanura (silverfish).
Table 4.3 Invertebrate captures during all surveys and methods between November 2012 and November 2014. Detection rate is the proportion of times the species was detected across all sites and survey periods. Bold indicates species with sufficient number of captures for individual abundance modelling.

<table>
<thead>
<tr>
<th>Order</th>
<th>Common Name</th>
<th>Captures</th>
<th>Detection rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aranae</td>
<td>Spiders</td>
<td>1 670</td>
<td>1.00</td>
</tr>
<tr>
<td>Blattodea</td>
<td>Cockroaches</td>
<td>236</td>
<td>1.00</td>
</tr>
<tr>
<td>Chilopoda</td>
<td>Centipedes</td>
<td>207</td>
<td>1.00</td>
</tr>
<tr>
<td>Coleoptera</td>
<td>Beetles</td>
<td>1 555</td>
<td>1.00</td>
</tr>
<tr>
<td>Collembola</td>
<td>Springtails</td>
<td>14</td>
<td>0.50</td>
</tr>
<tr>
<td>Dermaptera</td>
<td>Earwigs</td>
<td>4</td>
<td>0.17</td>
</tr>
<tr>
<td>Diptera</td>
<td>Flies</td>
<td>36</td>
<td>0.92</td>
</tr>
<tr>
<td>Embioptera</td>
<td>Web spinners</td>
<td>3</td>
<td>0.17</td>
</tr>
<tr>
<td>Hemiptera</td>
<td>True bugs</td>
<td>191</td>
<td>1.00</td>
</tr>
<tr>
<td>Hymenoptera (Ants only)</td>
<td>Ants</td>
<td>44 839</td>
<td>1.00</td>
</tr>
<tr>
<td>Hymenoptera (not Ants)</td>
<td>Bees/Wasps</td>
<td>8</td>
<td>0.42</td>
</tr>
<tr>
<td>Isopoda</td>
<td>Slaters</td>
<td>523</td>
<td>1.00</td>
</tr>
<tr>
<td>Isoptera</td>
<td>Termites</td>
<td>675</td>
<td>1.00</td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Moths/Butterflies</td>
<td>6</td>
<td>0.25</td>
</tr>
<tr>
<td>Mantodea</td>
<td>Mantids</td>
<td>3</td>
<td>0.25</td>
</tr>
<tr>
<td>Mecoptera</td>
<td>Scorpion flies</td>
<td>1</td>
<td>0.08</td>
</tr>
<tr>
<td>Megaloptera</td>
<td>Alderflies/Dobsonflies</td>
<td>1</td>
<td>0.08</td>
</tr>
<tr>
<td>Neuroptera</td>
<td>Lacewings</td>
<td>8</td>
<td>0.42</td>
</tr>
<tr>
<td>Odonata</td>
<td>Dragonflies/Damselflies</td>
<td>3</td>
<td>0.25</td>
</tr>
<tr>
<td>Orthoptera</td>
<td>Grasshoppers/Crickets</td>
<td>460</td>
<td>1.00</td>
</tr>
<tr>
<td>Phasmatodea</td>
<td>Stick/Leaf Insects</td>
<td>19</td>
<td>0.67</td>
</tr>
<tr>
<td>Phthiraptera</td>
<td>Lice</td>
<td>6</td>
<td>0.50</td>
</tr>
<tr>
<td>Plecoptera</td>
<td>Stoneflies</td>
<td>5</td>
<td>0.17</td>
</tr>
<tr>
<td>Psocoptera</td>
<td>Booklice</td>
<td>2</td>
<td>0.08</td>
</tr>
<tr>
<td>Scorpionidae</td>
<td>Scorpions</td>
<td>95</td>
<td>1.00</td>
</tr>
<tr>
<td>Thysanoptera</td>
<td>Thrips</td>
<td>7</td>
<td>0.25</td>
</tr>
<tr>
<td>Thysanura</td>
<td>Silverfish</td>
<td>195</td>
<td>1.00</td>
</tr>
</tbody>
</table>
Examination of the relationships between orders indicated 21 significant positive relationships between the orders tested (Table 4.4). However, the overall strength of the relationships was weak (coefficients < 0.4). The exception was the moderately strong relationships between the orders Chilopoda, Aranae and Orthoptera (Evans 1996).

Table 4.4 Spearman's rank-order correlations between invertebrate ordinal abundance (# individuals) across all surveys. Bold indicates significance; * P < 0.05, ** P < 0.01, *** P < 0.001. Underline indicates correlation coefficients > 0.4.

<table>
<thead>
<tr>
<th></th>
<th>Aranae</th>
<th>Blattodea</th>
<th>Chilopoda</th>
<th>Coleoptera</th>
<th>Ants</th>
<th>Isopoda</th>
<th>Orthoptera</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blattodea</td>
<td></td>
<td><strong>0.34</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chilopoda</td>
<td><strong>0.54</strong></td>
<td><strong>0.29</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coleoptera</td>
<td><strong>0.29</strong></td>
<td><strong>0.21</strong></td>
<td><strong>0.30</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ants</td>
<td><strong>0.25</strong></td>
<td>-0.05</td>
<td><strong>0.24</strong></td>
<td><strong>0.32</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Isopoda</td>
<td>0.11</td>
<td>0.09</td>
<td><strong>0.27</strong></td>
<td><strong>0.23</strong></td>
<td><strong>0.29</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Orthoptera</td>
<td><strong>0.49</strong></td>
<td><strong>0.38</strong></td>
<td><strong>0.56</strong></td>
<td><strong>0.39</strong></td>
<td><strong>0.29</strong></td>
<td>0.13</td>
<td></td>
</tr>
<tr>
<td>Thysanura</td>
<td><strong>0.21</strong></td>
<td>0.16</td>
<td>0.12</td>
<td>0.09</td>
<td><strong>0.20</strong></td>
<td><strong>0.21</strong></td>
<td><strong>0.22</strong></td>
</tr>
</tbody>
</table>

**4.3.2. Vegetation, climate, vertebrate predators and fire**

Two components of the PCA analysis explained 88.73% of variation between sites and surveys. In the two components only three variables; Triodia, bare ground and shrubs, strongly contributed to the differences between sites and surveys (loadings > 0.3). Unsurprisingly, the level of Triodia cover had a strong negative correlation with increasing fire extent and bare ground. Therefore, only Triodia and shrubs were included in the final models to represent changes in fire and vegetation.
Over time rainfall, temperature and predator presence varied considerably. The two large predators, cats and wild dogs, showed an inverse relationship until February 2014 after which time both increased steadily until the completion of the study (See Chapter 3). The abundance of small predators varied considerably over time and between sites (Figure 4.1). Reptiles showed the expected seasonal fluctuation that reflected their ectothermic requirements, with low capture rates in the cooler periods (May and August) and higher capture rates in the warmer month (November and February). Greater variation in reptile abundances between sites was also found in the warmer months. Until August 2014, small mammal abundance was generally low with little variation between sites. During August and December 2014, abundance and variation between individual sites increased dramatically.

Larger than normal monthly rainfall (over 100 mm) was recorded in May 2013 and January 2014. Temperature varied as expected with low monthly temperatures in the winter months (May and August) and high monthly temperatures recorded in the summer months (November and February; See Chapter 3).
4.3.3. Invertebrate response

The number of models that showed considerable support ($\Delta$AIC$_c < 4$) varied from one to six models for all response variables tested (Table 4.5; Appendix 5). I selected the top ranked model as the ‘best fit’ model for Aranae, Chilopoda, Coleoptera, Hymenoptera, Isopoda and Thysanura, as all other models showed considerably lower Akaike weights and did not substantially increase the deviance explained by the model. I selected the first and fourth ranked models as the best fit models for invertebrate richness as the increased complexity in the models ranked two and three (brought on by additional variables) did not considerably improve model weight or the deviance explained by the model. The top two models were selected as equal ‘best fit’ for Blattodea abundance, as both showed similar Akaike weights and explained similar levels of null deviance. Only a single model was considered a good fit from all Orthoptera candidate models. Notably, even the ‘best fit’ models for invertebrate richness and
Blattodea abundance explained relatively low amounts of the deviance in the data (invertebrate richness; 39.70% and 41.02% and Blattodea abundance; 34.35% and 35.89%).

Table 4.5 Model selection summary statistics of invertebrate GAMMS with considerable support (ΔAICc < 4). Model covariates indicate variable groups as specified in Table 4.2. df = degrees of freedom in model, AICc = Akaike’s Information Criterion corrected for small sample size, ΔAICc = difference from lowest AICc in candidate models, wi = Akaike weight of model and % Dev = percentage null deviance explained by model. Models considered ‘best fit’ model are underlined.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>Rank – Model covariates</th>
<th>df</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>wi</th>
<th>% Dev</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Invertebrate richness</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Long term climate</td>
<td>5</td>
<td>473.62</td>
<td>0.00</td>
<td>0.25</td>
<td>39.70</td>
<td></td>
</tr>
<tr>
<td>2 – Lrg predators + Long term climate</td>
<td>5</td>
<td>473.69</td>
<td>0.07</td>
<td>0.24</td>
<td>40.75</td>
<td></td>
</tr>
<tr>
<td>3 – Sml predators + Long term climate</td>
<td>5</td>
<td>474.01</td>
<td>0.38</td>
<td>0.21</td>
<td>40.52</td>
<td></td>
</tr>
<tr>
<td>4 – Lrg predators + Short term climate</td>
<td>5</td>
<td>474.05</td>
<td>0.42</td>
<td>0.21</td>
<td>41.02</td>
<td></td>
</tr>
<tr>
<td><strong>Aranae</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Lrg predators + Long term climate</td>
<td>17</td>
<td>732.92</td>
<td>0.00</td>
<td>0.44</td>
<td>56.22</td>
<td></td>
</tr>
<tr>
<td>2 – Vegetation + Short term climate</td>
<td>15</td>
<td>734.06</td>
<td>1.13</td>
<td>0.25</td>
<td>53.25</td>
<td></td>
</tr>
<tr>
<td>3 – Short term climate</td>
<td>14</td>
<td>735.79</td>
<td>2.86</td>
<td>0.11</td>
<td>50.48</td>
<td></td>
</tr>
<tr>
<td>4 – Lrg predators + Short term climate</td>
<td>19</td>
<td>735.91</td>
<td>2.99</td>
<td>0.10</td>
<td>57.25</td>
<td></td>
</tr>
<tr>
<td>5 – Sml predators + Short term climate</td>
<td>17</td>
<td>736.23</td>
<td>3.30</td>
<td>0.08</td>
<td>54.07</td>
<td></td>
</tr>
<tr>
<td><strong>Blattodea</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Lrg predators + Long term climate</td>
<td>10</td>
<td>401.70</td>
<td>0.00</td>
<td>0.27</td>
<td>34.35</td>
<td></td>
</tr>
<tr>
<td>2 – Sml predators + Long term climate</td>
<td>12</td>
<td>402.27</td>
<td>0.57</td>
<td>0.20</td>
<td>35.89</td>
<td></td>
</tr>
<tr>
<td>3 – Long term climate</td>
<td>8</td>
<td>402.51</td>
<td>0.81</td>
<td>0.18</td>
<td>30.10</td>
<td></td>
</tr>
<tr>
<td>4 – Vegetation + Short term climate</td>
<td>12</td>
<td>403.88</td>
<td>2.18</td>
<td>0.09</td>
<td>35.64</td>
<td></td>
</tr>
<tr>
<td>5 – Sml predators + Short term climate</td>
<td>9</td>
<td>403.90</td>
<td>2.20</td>
<td>0.09</td>
<td>30.51</td>
<td></td>
</tr>
<tr>
<td>6 – Short term climate</td>
<td>7</td>
<td>404.45</td>
<td>2.77</td>
<td>0.07</td>
<td>27.02</td>
<td></td>
</tr>
<tr>
<td>7 – Lrg predators + Short term climate</td>
<td>9</td>
<td>404.93</td>
<td>3.23</td>
<td>0.05</td>
<td>30.54</td>
<td></td>
</tr>
<tr>
<td><strong>Chilopoda</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 – Sml predators + Short term climate</td>
<td>23</td>
<td>308.11</td>
<td>0.00</td>
<td>0.73</td>
<td>78.92</td>
<td></td>
</tr>
<tr>
<td>2 – Vegetation + Long term climate</td>
<td>21</td>
<td>312.08</td>
<td>3.97</td>
<td>0.10</td>
<td>76.11</td>
<td></td>
</tr>
</tbody>
</table>
### Coleoptera

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>AIC</th>
<th>edf</th>
<th>p</th>
<th>R^2</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 – Lrg predators + Short term climate</td>
<td>15</td>
<td>703.94</td>
<td>0.00</td>
<td>0.79</td>
<td>59.15</td>
</tr>
<tr>
<td>2 – Short term climate</td>
<td>15</td>
<td>707.01</td>
<td>3.07</td>
<td>0.17</td>
<td>57.35</td>
</tr>
</tbody>
</table>

### Hymenoptera

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>AIC</th>
<th>edf</th>
<th>p</th>
<th>R^2</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 – Lrg predators + Long term climate</td>
<td>19</td>
<td>1360.13</td>
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### Isopoda

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<th>edf</th>
<th>p</th>
<th>R^2</th>
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### Thysanura

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Across all invertebrate GAMMs, the variable group, vegetation, was not included in any of the ‘best fit’ models (Figure 4.2, Table 4.5). The presence of large predators was included in all ‘best fit’ models except Chilopoda and Isopoda. Increased cat presence across sites was only considered a significant positive predictor for invertebrate richness (4th ranked model; edf = 1.25, p = 0.01) and Orthoptera abundance (edf = 1.95, p < 0.001). Similarly increased wild dog presence was considered a consistent positive predictor of Aranae (edf = 1.00, p = 0.001) and Blattodea (top ranked model; edf = 1.00, p = 0.03) abundances. Wild dog presence was also a significant predictor for Ant abundance (edf = 2.99, p = 0.02) but the relationship fluctuated with ant abundance being the lowest when wild dog presence was recorded at 45% of sites.
and the highest when wild dog presence was recorded at 70% of sites. Small predators were only found to be a significant predictor of Chilopoda abundance which increased with greater abundances of both mammals (edf = 1.00, p = 0.002) and reptiles (edf = 6.55, p = 0.006).

Figure 4.2 Smoothing plots for non-climate variables present in top GAMMS for invertebrates. Solid line = GAMM model fit; dotted line indicates 95% confidence bands; plots of significant variables (p > 0.05) are shown in bold. Missing plots indicate variables not present in top models. Black indicates top ranked model; Grey indicates second ranked model.
Overall, climate variables were clearly the most consistent predictors across invertebrate richness and individual ordinal abundances (Figure 4.3). Minimum temperatures were positive predictors for Isopoda (edf = 1.00, p < 0.001), Orthoptera (edf = 1.00, p = 0.005) and invertebrates richness (top ranked model; edf = 1.14, p < 0.001) over one day, 13 days and 12 months, respectively. The average minimum temperature recorded over 14 days and 1 month prior was also a significant predictor for Coleoptera (edf = 3.84, p = 0.03) and Aranae (edf = 4.73, p < 0.001) abundance, respectively. Both abundances fluctuated with increasing temperature but in opposite directions. When temperatures reached 12 °C Coleoptera abundance peaked while Aranae abundance dropped and when temperatures reached 20 °C Aranae peaked and Coleoptera dropped. Minimum temperature over the previous month was also a good predictor for Blattodea abundance (top ranked model; edf = 2.66, p < 0.001; second ranked model; edf = 2.89, p = 0.008) with greater abundance recorded when average temperatures exceeded 15 °C. Maximum temperatures were positive predictors for ants (edf = 1.00, p < 0.001) and Thysanuran abundance (edf = 5.00, p < 0.001) over 13 months and 14 days, respectively.

The number of rain days over short time periods (6 – 10 days prior) were considered significant positive predictors of invertebrate richness (4th ranked model; edf = 1.00, p < 0.001), Chilopoda abundance (edf = 1.00, p < 0.001), Coleoptera abundance (edf = 1.00, p < 0.001) and Orthoptera abundance (edf = 1.00, p < 0.001). The number of rain days within 15 days prior to sampling was also a significant predictor of Isopoda abundance (edf = 2.86, p < 0.001); however, abundance decreased when two or three rain days were recorded yet increased dramatically when the number of rain days exceeded three. The number of rain days within one month of sampling was a positive predictor of
Araeae abundance ($edf = 1.00, p < 0.001$) and Blattodea abundance once rain
days exceeded five (top ranked model; $edf = 2.71, p = 0.01$; second ranked model;
$edf = 2.64, p = 0.09$). The total rainfall 13 months prior to sampling was also a
positive predictor for ant abundance ($edf = 1.00, p < 0.001$).

Figure 4.3 Smoothing plots for climate variables present in top GAMMS for
invertebrates. Solid line = GAMM model fit; dotted line indicates 95% confidence bands;
plots of significant variables ($p > 0.05$) are shown in bold. Missing plots indicate
variables not present in top models. Black indicates top ranked model; Grey indicates
second ranked model.
4.4. Discussion

The results of this study contribute substantially to the information available on invertebrate occurrence in Australia’s arid zone. Our study sampled 25 orders with many occurring relatively infrequently in samples, highlighting the diversity in the area and potential difficulty in gaining information uniformly across all orders. Deploying a greater diversity of trap methods may improve sampling across orders and is recommended for studies in the future. Overall capture rates of orders partially coincided with the composition of invertebrate assemblages seen in other studies (e.g. Gibson 2001; Kwok & Eldridge 2015; Kwok et al. 2016; Teasdale et al. 2013). My results support existing work that show high numbers of Aranae and Coleoptera and the clear dominance of ants in invertebrate samples from sandy deserts (e.g. Kwok et al. 2016) and other arid regions in Australia (Kwok & Eldridge 2015; Pavey et al. 2016). Interestingly, a study by Kwok et al. (2016), with similar methods and conducted in a similar habitat, recorded low Collembola and high Isopoda abundances. In the Kwok et al. (2016) study, Collembola abundance was high during and immediately following rainfall but ‘declined sharply and remained relatively low’ during periods of low rainfall, which was the period of interest in our study. Although during this study several larger rainfall events occurred, they may not have been sufficient to promote Collembola abundance. Furthermore, rainfall is likely a factor influencing the higher number of Isopoda in this study. Isopoda are known to prefer moist environments and the relatively high mean annual rainfall at our site (334.2 mm) may support populations more readily. The potential competitive interaction between the two detritivores, Collembola and Isopoda, may have influenced abundances. If greater rainfall supports more stable populations of Isopoda they may compete for resources and suppress
Collembola abundance, similar to the competition found in freshwater detritivores (Basset & Rossi 1990). Further investigation into the relationship between Collembola and rainfall thresholds and the potential competitive suppression between orders in arid environments would help clarify this.

Of the orders examined here, the results showed mixed support for the initial hypotheses. The predictions regarding the influence of short-term temperature and rainfall were supported in most orders; however, there was no support for my prediction that climate thresholds would be apparent. Vegetation was also not found to influence the richness and abundance of invertebrate orders. Although large predators were found to be an important predictor for many orders, the relationship was positive and therefore not as expected. My predictions regarding the interactions between invertebrate orders also found little support in the results. Overall, the findings indicated that during a period post boom invertebrate richness and abundance was predominantly driven by a combination of short and long-term climate variables.

4.4.1. Invertebrate response to climate variables
Climate variables were important predictor variables for all invertebrate orders tested. Specifically, minimum temperatures (within 1 month of sampling) were an important predictor for most of the orders tested. These findings support many previous studies that have shown positive relationships between invertebrates and temperature (e.g. Kwok et al. 2016; Langlands et al. 2006; Palmer 2010; Schultheiss & Nooten 2013). The fluctuating response to temperature by Aranae and Coleoptera here is likely a seasonal variation in activity. The former being active during temperature extremes (summer and winter) whilst the latter is active in the more temperate periods between. Previous studies have shown Aranae species assemblages varying considerably
across sampling months (Langlands et al. 2006), therefore the fluctuations in this study are likely due to species specific responses to seasonal fluctuations. Furthermore, the greater activity of Aranae species during periods when conditions are physiologically sub-optimal (winter) would potentially reduce competition and create a partitioning of resources. Several species (namely Embioptera and wasps) are known to be active during winter periods in arid Australia (Palmer 2010) and therefore could help to support the survival of invertebrate predators during these periods. Although many previous studies have shown Coleoptera abundance to be highest in warmer months (Blanche et al. 2001; Driscoll & Weir 2005), rainfall has also been found to be a strong driver of abundance (this study; Kwok et al. 2016). The rainfall events during our study may have created pulses in activity that superseded the expected seasonal influences.

Overall invertebrate richness and ant abundance did follow seasonal patterns, although were more strongly related to the temperatures of the previous season (12 and 13 months prior, respectively). The rate of invertebrate development and growth is closely linked to higher temperatures (Bale et al. 2002) and therefore greater temperatures the previous season may have decreased generation times, increased breeding periods and promoted greater breeding success across a variety of species. The breeding of many invertebrates involves many stages of development; eggs, larvae, nymphs, pupa, adult, etc. (Zborowski & Storey 2010). This development can occur quickly or take several months, with the ability to detect individuals usually greatest only at the completion of development (Zborowski & Storey 2010). As activity is also highly regulated by seasonal fluctuations, an increase in breeding success was likely to be more detectable across a wide variety of species the following season once
development is complete and temperatures are optimal. Specifically, the development of social species, such as ants, usually occurs in a communal nest with young being cared for by colony ‘workers’ during development (Wilson 1983). Therefore, increased breeding is unlikely to be detected until young become active the following season.

With the exception of Thysanura, greater rainfall was an important positive predictor of abundance for all orders tested which coincides with many previous studies on invertebrates (Barrow et al. 2007; Blanche et al. 2001; Hunter & Elder 1999; Letnic et al. 2004; Palmer 2010). The lag in the response of ant abundance (to 13 months) was found and likely due an increase in successful reproduction the previous season, as described above in relation to temperature.

Surprisingly, our results failed to detect a positive association between Thysanura and rainfall. Thysanura are known to withstand some desiccation by absorbing water directly from the atmosphere (Smith & Watson 1991) therefore the ‘top up’ rainfalls experienced during this study may have buffered populations against the expected negative response through increased levels of humidity.

4.4.2. Invertebrate response to vegetation
I found no support for my predictions regarding the importance of vegetation cover to invertebrate richness and ordinal abundance. In a similar study in the Simpson desert the degree of cover provided by non-\textit{Triodia} species was found to be the most consistent predictor of invertebrate abundance (Kwok et al. 2016). My study showed relatively little variation in the amount of cover provided by non-\textit{Triodia} species spatially or temporally, whilst Kwok et al. (2016) noted large fluctuations (likely due to the longer time periods sampled). Kwok et al.
(2016) also found flowering and seed production to be an important factor in predicting invertebrate abundance, which was not included in models here. More detailed examination of vegetation characteristics between sites, particularly regarding the level of seeding/flowering, would improve the informative capabilities of future studies.

4.4.3. *Invertebrate response to predation*

The results here showed mixed support for my hypotheses regarding predation. In support of predictions, small native predators (mammals and reptiles) were only considered a significant predictor for Chilopoda abundance and were never seen to create detrimental top down effect. As a predatory order, Chilopoda are known to consume small mammals and reptiles (e.g. MCC 2003) and therefore it is not surprising they show a positive response to increases in their abundance.

Although I predicted greater presence of large mammalian predators would negatively affect invertebrate richness and abundance, the opposite relationship was found. Orthoptera abundance increased with greater cat presence and Aranae and Blattodea responded positively to increased wild dog presence. A recent examination of cat diets in similar habitats during periods of boom and bust showed a dramatic increase in the consumption of invertebrates during the bust period as preferred prey items (small mammals and reptiles) became less abundant (Yip et al. 2015). As Orthoptera is one of the dominant invertebrate orders in the area, it is likely they are a key component of cat diets during periods of lower rainfall. In comparison, although dingoes (the dominant wild dog species in the region) are known to readily eat invertebrates (Allen et al. 2012) previous studies have shown invertebrates are not an important component of wild dog diets locally (e.g. Paltridge 2002; Spencer et al. 2014). Due to the inverse relationship between the two large predators at my study site
populations of Aranae and Blattodea may be more likely to survive when wild
dogs are more abundant and potentially play a suppressive role on cat
abundance (Cremona et al. 2014; Fleming et al. 2012; Letnic et al. 2012). The
overall lack of negative influence of large predators may also be due to the
overall favourable climatic conditions in the region. High temperatures
combined with generally greater annual rainfall may help to buffer invertebrate
populations from the pressures imposed by large predators in the region.
Alternatively, the relationships found here might also be due to communal
seasonal responses of invertebrates, cats and wild dogs in the area. The strong
relationships found between most invertebrate orders and seasonal fluctuations
in large predators (previously discussed in Chapter 3) would likely have a
considerable influence on the strength of the drivers found here.

Overall, the results of this study provide a broad assessment of invertebrate
relationships in arid *Triodia* grasslands. Although I found several strong
patterns in the response of invertebrates to environmental factors, many of the
‘best fit’ models in this study explain less than 70% of the variation in data
indicating that potentially important variables remain untested. Invertebrates
play a key role in the functioning of ecosystems from decomposers to producers
to consumers. Gaining a better understanding of what drives their persistence is
essential. Understanding their roles and relationships in the ecosystem and
ensuring their stability is an important (and largely forgotten) step in managing
systems. I recommend greater monitoring of invertebrates in arid zone
management, as the flow on effects to changes in abundance and diversity could
be substantial. Furthermore, given climate change predictions indicate boom
and bust cycles are likely to become more intense across Australian arid zone
(Intergovernmental Panel on Climate Change [IPCC] 2014) monitoring
invertebrate assemblages may become increasingly important in maintaining ecosystems into the future.
CHAPTER 5

REFUGE USE IN THE BRUSH-TAILED MULGARA
(DASYCERCUS BLYTHI)

Top to bottom: Newhaven Wildlife Sanctuary (J Molyneux); Dasycercus blythi (J Schofield/AWC); Spinifex sand plains (J Molyneux)
Abstract
The brush-tailed mulgara (*Dasycercus blythi*) is the largest remaining dasyurid in central Australia and is considered an important species in Australia's arid zone. Although previous studies have indicated *D. blythi* increase in abundance during periods of high rainfall, little is known regarding the spatial expansion and contraction that is likely to occur with these fluctuations. This study examined long-term trapping and sign survey records of *D. blythi* across two locations in the Northern Territory; Newhaven Wildlife Sanctuary and Uluru Kata-Tjuta National Park. I examined the level of detection and extent of *D. blythi* occurrence across both locations and how this related to variations in rainfall during the previous year. I found *D. blythi* showed an increased level of detection when rainfall increased above 400 mm the previous year. When rainfall was below this threshold, *D. blythi* showed contractions to core areas in both sites, although contraction was more pronounced at Uluru Kata-Tjuta National Park. I further assessed how the presence of fire and timing of rainfall in relation to reproductive stages may influence detection of *D. blythi* during periods of lower rainfall. Overall, my results indicated fire had little effect, while greater rainfall prior to weaning resulted in greater detection of *D. blythi* the following year. These results provide greater insight into how *D. blythi* has survived across its broad distribution while similar species have gone extinct in the region. It helps to refine management areas and through greater knowledge increase the ability of land managers to support populations into the future.
5.1. Introduction

The distribution and abundance of small mammals in arid environments can fluctuate dramatically. Previous research has shown that many species have irruptive population dynamics. Massive increases in population sizes occur when conditions are favourable followed by dramatic declines when conditions deteriorate (Letnic & Dickman 2006; Letnic et al. 2005). The dominant driving influence in arid systems for these fluctuations is rainfall (Morton et al. 2011). Higher rainfall drives pulses in primary production that flow through trophic pathways via greater resource availability and/or reproductive success (Letnic et al. 2011; Letnic et al. 2005; Pavey & Nano 2013a). Although pulses in populations are well documented in Australian rodents (e.g. Letnic et al. 2005; Pavey & Nano 2013b; Pavey et al. 2014; Predavec 1994) little is known of similar trends in Australia’s other dominant terrestrial small mammal group, the dasyurid marsupials.

The high number of dasyurid species inhabiting arid environments is unique to Australia (Byrne et al. 2008). With relatively fixed life-history strategies (i.e. seasonal and/or monoestrous breeding) this groups’ ability to rainfall pulses is considered potentially restricted (Dickman et al. 2001). Unlike rodents in the region, who are able to reproduce quickly throughout the year, dasyurids would likely only respond when rainfall occurs at specific times that would improve reproductive success. Moreover, our understanding of population fluctuations has been limited as many species have relatively low detection rates, particularly when populations are at low densities. Furthermore, given their lack of flexibility in breeding, the ability of species to respond to rainfall may be limited and dependent on rainfall occurring during specific periods that promote the greatest reproductive success (Goldman 1999).
*Dasycercus blythi* is one of the largest (50 – 120g) remaining dasyurids in the sand plains of western and central Australia (Van Dyck et al. 2008). Invertebrates are the predominant prey of choice with selection of invertebrate orders varying between populations and seasons (Chen et al. 1998; Masters 1998). Although not as prevalent in the diet as invertebrates, mammals, reptiles and birds are consistently consumed by *D. blythi* (Chen et al. 1998). During periods of high resource availability *D. blythi* store fat in their tails and can fluctuate dramatically in body weight and condition depending on resource availability and breeding requirements (Masters 1998). Individuals can enter daily torpor to reduce energetic requirements which further aids in maintaining stable home ranges when resources are scarce (Chen et al. 1998; Körtner et al. 2008).

The density and distribution of *D. blythi* populations are largely driven by longer-term climatic conditions (El Niño-Southern Oscillation; ENSO). However, they also experience smaller scale annual declines during the breeding season (May – September) followed by an increase caused by the emergence of juveniles into the population (Masters & Dickman 2012). The successful persistence of *D. blythi* in the region compared to many other similar species (i.e. kowari, red-tailed phascogale) is considered to be a result of a flexible diet, ability to enter torpor and ability to withstand large fluctuations in body weight (Masters & Dickman 2012).

Although *D. blythi* populations have shown declines since European settlement, they are currently considered stable. Currently, small low density isolated populations are apparently able to persist across the landscape during periods of low rainfall (Woinarski et al. 2014). In the mid-1990s areas of prime habitat for *D. blythi* were identified as areas with heavy sandy soils, paleo-drainage
influences (areas surrounding paleo drainage lines; i.e. historical watercourses such as rivers or streams that no longer flow), mature *Triodia* hummocks separated by open ground and the presence of shrubs (likely indicating greater ground water availability; Baker 1996). More recently *D. blythi* have also been recorded across a wide a variety of habitats, including gibber plains, woodlands and gypsum lakes (AWC 2016a; Pavey et al. 2011). Detectability of *D. blythi* is variable, with populations regularly disappearing from monitoring programs during periods of low rainfall (e.g. Dickman et al. 2001). The ability of populations to re-establish across a broader landscape when conditions are favourable indicates populations persist in the landscape, likely in areas of suitable habitat that enable expansion when conditions improve (Dickman et al. 2011; Gibson & Cole 1992; Letnic et al. 2011). The areas of prime habitat identified previously are usually vast and little information available on how populations are distributed within them when abundances are at their lowest and most vulnerable. Gaining this knowledge is important in order to manage populations effectively when conditions are least favourable and the threat of declines most intense.

Historically, the primary driver of population declines in *D. blythi* was considered to be changes in fire regimes (Baker 1996; Masters 1998; Masters et al. 2003). This has resulted in the extensive use of fire management in areas supporting *D. blythi* populations. However, recent studies have shown *D. blythi* readily persist in recently burnt areas (Thompson & Thompson 2007a) obscuring our understanding of the factors that influence *D. blythi* distribution and persistence.

This study will focus on two areas considered to be prime *D. blythi* habitat (as described by Baker 1996) that have been intensively managed through specific
fire strategies in recent years. Utilising data collected across several years I will examine how detection of *D. blythi* varies in prime habitat and surrounding areas in relation to annual rainfall on a broad scale. I examined the spatial patterns of *D. blythi* persistence within prime habitat during periods of low rainfall. I will further explore whether fire and the timing of rainfall in relation to reproduction can help to predict the occurrence of *D. blythi* during drought.

Overall, I predict:

1. Detection of *D. blythi* will be greater when higher annual rainfall is recorded the previous year.
2. During periods of lower rainfall, key areas within prime habitat will support *D. blythi* populations more consistently; however, sporadic detection will still be apparent across the broader area.
3. *D. blythi* will be more readily detected in areas unaffected by recent fire and when rainfall the previous year coincides with the weaning and dispersal of young.

### 5.2. Methods

#### 5.2.1. Study sites and rainfall data

I examined *Dasycercus blythi* occurrence data across two locations in the Northern Territory; Newhaven Wildlife Sanctuary and Uluru Kata-Tjuta National Park (UKJNP). Methods data were obtained from Newhaven’s general fauna surveys between 2008 and 2015 and *D. blythi* targeted monitoring undertaken in UKJNP between 2000 and 2011. Monthly rainfall totals have been collected from the Newhaven homestead since 1962 and Yulara airport since 1985 (See Chapter 2 for details).
Chapter 5 | Refuge use

5.2.2. *Dasycercus blythi* sampling

5.2.2.1. Newhaven Wildlife Sanctuary

At Newhaven 171 sites were surveyed from 2008 until 2015. Sites were surveyed annually (between March and May) by one of two methods; live trapping or sign surveys. Not all sites were surveyed every year due to accessibility and logistical constraints (Table 5.1; Figure 5.1).

Table 5.1 Summary of trapping and sign surveys conducted at Newhaven Wildlife Sanctuary between 2008 and 2015.

<table>
<thead>
<tr>
<th>Year</th>
<th>Number of Live Trapping sites</th>
<th>Number of Sign Surveys sites</th>
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</tr>
<tr>
<td>2015</td>
<td>56</td>
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</table>

Figure 5.1 Location of live trapping (closed) and sign survey (open) sites at Newhaven Wildlife Sanctuary between 2008 and 2015.
Live trapping and sign survey sites were each stratified across 15 habitat types described by Latz et al. (2003) in a detailed vegetation assessment conducted across the property. Site locations were further selected to represent varying distances to water in each habitat type and varying ages of fires history (0 to > 25 years since fire; Moore 2013). At each site live trapping surveys were conducted annually in April over three consecutive nights. Due to unsuitable terrain for pitfall trapping, sites situated on rocky ranges were surveyed using box traps across four transects situated approximately 200 m apart. Each transect consisted of 25 box traps (type A: 30 x 10 x 8 cm; Elliott Scientific Equipment, Upwey, VIC, Australia) running from the base of the range to the top with approximately 10 m between consecutive traps. At all remaining sites an area 100 x 100 m was surveyed with 20 box traps, 12 pitfall traps and 12 shade-mesh funnel traps (Appendix 6; 18 x 18 x 75 cm; Terrestrial Ecosystems, Claremont, WA, Australia). Box traps were spaced 20 m apart following the perimeter of the site and baited with a standard bait of peanut butter and oats. Box traps were checked early morning, closed during the day and opened late afternoon. Pitfalls were deployed across six lines each consisting of two PVC pipes situated approximately 30 metres apart (16 cm diameter, 60 cm deep), buried flush to the ground and connected by a dampcourse drift fence (30 cm high) that continued a further 2.5 metres on either side. The six pitfall lines were placed in three evenly spaced parallel transects (two lines per transect) across the trapping area. Funnel traps were centrally located on each side of the drift fence and insulated to reduce heat stress. Both pitfall traps and funnel traps were checked early morning and late afternoon and remained open for the entire survey period. All trapping equipment was removed from the sites between surveys, except PVC pipes, which were capped and mounded with sand
to prevent animals being caught between survey periods. All *D. blythi* caught were weighed, sexed and their reproductive condition recorded. To identify within survey recaptures fur on the rump of all individuals was clipped.

At Newhaven, sign surveys were also conducted annually across a 2 ha site (100 m x 200 m) at each of the sign survey sites. Each site was searched for tracks, scats and other signs for 0.5 person hours between sunrise and 11am to ensure optimal conditions (Moore 2013). Each sign was classified as fresh (< 2 days old) or old (> 2 days old) and were predominantly completed by highly experienced local trackers. All surveys were conducted following 2 days of optimal weather (e.g. no wind or rain) to eliminate varying weather conditions creating bias within the data.

5.2.2.2. **Uluru Kata–Tjuta National Park**

In the Uluru Kata–Tjuta National Park (UKJNP) a total of 124 sites were surveyed between 2000 and 2011. In June 2000, sign surveys were conducted across 32 sites stratified evenly over four spinifex dominated habitat categories; *Triodia basedowii* sandplain, *T. basedowii* dunefield, *T. pungens* sandplain and *T. pungens* dunefield (Appendix 7). Specific site locations focused predominantly on the borefields area (See Chapter 2 for details), were selected randomly from easily accessible areas in the park and followed the trapping protocols for *D. blythi* as proposed by Masters (1997) that required a minimum of 1 km distance between sites. The majority of sites (72%) were last burnt in 1976 whilst the remaining sites were burnt between nine to 16 years prior. At each site a 300 m x 40 m transect was searched for any sign of *D. blythi* (including diggings, burrows, tracks and scats) by a highly experienced senior Anangu (Indigenous landowners).
Between November 2000 and November 2008 live trapping surveys were conducted across 10 sites. Sites were located in the central borefields area in UKJNP. Exact site locations varied depending on the occurrence of fire the previous year, with many sites being relocated to areas of unburnt vegetation to ensure every survey sampled across all fire histories (Appendix 7). All sites were trapped annually in November for three consecutive nights with 25 box traps (type A: 30 x 10 x 8 cm; Elliott Scientific Equipment, Upwey, VIC, Australia) and baited with peanut butter and oats. Box traps were laid out 30 m apart in a horseshoe pattern covering approximately 6 ha. All *D. blythi* caught were weighed, sexed and tail width (generic measure of condition) was recorded. Individuals were marked at the base of the tail with a felt pen to identify recaptures within a survey period. In 2008, additional sign surveys were conducted at 20 sites across the UKJNP. Site locations were selected randomly, in areas of suitable habitat. All sites covered an area of approximately 2.5 ha and were searched for 20 minutes by one experienced personnel who recorded any sign of *D. blythi* presence. From 2009 until 2011 box trapping continued but large variations in trap effort and design were utilised during this period (Table 5.2; Appendix 7).

Table 5.2 Trapping regime used for *D. blythi* surveys at Uluru Kata –Tjuta National Park between 2009 and 2011 (ADNP 2012).

<table>
<thead>
<tr>
<th>Date</th>
<th>Number of Sites</th>
<th>Number of traps per site</th>
<th>Number of trap nights</th>
<th>Area surveyed per site</th>
</tr>
</thead>
<tbody>
<tr>
<td>July 2009</td>
<td>10</td>
<td>25</td>
<td>4</td>
<td>0.5 ha</td>
</tr>
<tr>
<td>Sept 2009</td>
<td>1</td>
<td>250</td>
<td>4</td>
<td>9.4 ha</td>
</tr>
<tr>
<td>Aug 2010</td>
<td>2</td>
<td>250</td>
<td>4</td>
<td>5 ha</td>
</tr>
<tr>
<td>Oct 2011</td>
<td>4</td>
<td>100</td>
<td>3</td>
<td>6 ha</td>
</tr>
<tr>
<td>Nov 2011</td>
<td>12</td>
<td>25</td>
<td>4</td>
<td>2.25 ha</td>
</tr>
</tbody>
</table>
In addition to live trapping, sign surveys were conducted in March 2010 and March 2011 to assess \textit{D. blythi} occurrence across 18 randomly selected sites in UKJNP (Appendix 7). Each site covered an area of 4 ha and was searched systematically for 20 minutes by four personnel with moderate to high levels of experience.

\subsection*{5.2.3. \textit{Statistical analysis}}

\subsubsection*{5.2.3.1. Relationship between rainfall and \textit{D. blythi} detection}

In order to account for the varying number of sites sampled across years, I standardized the data by calculating the proportion of sites that positively detected \textit{D. blythi} for each year. From sign surveys, I used only recent sign (< 2 days old) as this was more comparative to the time period sampled by live trapping. Annual rainfall totals were calculated to represent ecologically relevant seasonal variations found in the region (summer dominated rainfall). Therefore, annual rainfall totals include the period between July of the previous year until June of the current year or the beginning of the relevant trapping period (whichever was earlier). Utilising this information I investigated the relationship between the proportion of sites where \textit{D. blythi} was detected and annual rainfall prior to sampling through generalised additive modelling (GAM) with a binomial distribution and logit link function (Zuur et al. 2014). Additive modelling was selected as initial data exploration indicated a non-linear pattern in the data. I included both rainfall and location (Newhaven and UKJNP) as covariates as I expected variations in the level of detection between locations due to the different sampling regimes (i.e. targeted vs general surveys). Given the limited number of replicates for each location (Newhaven, n = 8; UKJNP, n = 12) I was unable to estimate location specific responses. Therefore, I assumed the response of \textit{D. blythi} populations would be similar across location. Prior to
modelling all covariates were examined for outliers and collinearity using Pearson correlation coefficients (< 0.70) and variance inflation factors (< 3.00). All rainfall totals were standardised to avoid their larger measurement scales dominating correlation coefficients (Zuur et al. 2009). All modelling was conducted in mgcv package (Wood 2006) in the statistical program R (R Core Team 2015).

5.2.3.2. Spatial distribution of D. blythi occurrence during low rainfall
In order to assess refuge locations within prime habitat I extracted the data for all sites located within the preferred habitat of D. blythi, spinifex sand plains/dunes habitat. As I was only interested in periods of lower rainfall, I utilised the information from the previous analysis to indicate the level of rainfall at which D. blythi populations increased and discarded all years with rainfall above this. To ensure spatial independence between sites I grouped all sites (at each location) that occurred in 1 km of each other (henceforth referred to as g-sites). This distance was based on the reported daily average maximum distance moved by individual D. blythi in previous studies (Körtner et al. 2007; Körtner et al. 2016; Masters 2003). For each g-site, I then calculated either presence or absence of D. blythi in each year. G-sites were mapped using ArcGIS 10.1 (ESRI 2012) to visualise the frequency of D. blythi occurrence during periods of lower rainfall across spatial scales.

5.2.3.3. Factors influencing D. blythi occurrence during low rainfall
I further utilised the above absence/presence data to investigate the effect of fire and the timing of rainfall in relation to breeding during periods of low rainfall. For this analysis, I focused on the data from Newhaven as that provided more consistent measure of D. blythi occurrence across time and space. Based upon previous literature (Dickman et al. 2001; Masters 1998; Pavey et al. 2011) and
the breeding condition recorded for captured individuals at the site (See Chapter 6) rainfall totals were calculated across four reproductively relevant time periods (Table 5.3).

Table 5.3 Rainfall time periods utilised in relation to when D. blythi were sampled and explanation of the related ecological influence.

<table>
<thead>
<tr>
<th>Timing of rainfall</th>
<th>Ecological influence</th>
<th>Example rainfall period used for D. blythi sampled in March 2012</th>
</tr>
</thead>
<tbody>
<tr>
<td>Month prior to sampling</td>
<td>Short-term response</td>
<td>February 2012</td>
</tr>
<tr>
<td>Recent Summer (Nov – July or sampling month)</td>
<td>Survival of young post weaning</td>
<td>November 2011 until February 2012</td>
</tr>
<tr>
<td>Previous Summer (Nov – July)</td>
<td>Reproductive condition and success</td>
<td>November 2010 until July 2011</td>
</tr>
</tbody>
</table>

I recorded the number of D. blythi caught the previous season within each g-site to assess the consistency of breeding success within areas over time. To assess the effect of fire, I calculated the number of months since burn (minimum of 25% of the g-site area burnt) to a maximum of 300 months, as this reflects the estimated time period for Triodia to return to old growth status (Casson & Fox 1987). To assess patterns in relation to vegetation characteristics within prime habitat, I also included the broad vegetation classifications, Triodia pungens dominated plains and dunefields, and T. basedowii dominated plains and dunefields. Utilising this information, I ran generalised linear mixed effect models with a binomial distribution and a logit link function. I included g-site as a random variable to account for repeated sampling over time and trap effort (number of trap nights) as a covariate to account for varying sampling effort in individuals g-sites. Individual sign surveys were assessed as equivalent to 100 trap nights as this represented an approximate average trap effort of a standard
live trapping survey at Newhaven. All variables were checked for outliers and collinearity using Pearson correlation coefficients (< 0.70) and variance inflation factors (< 3.00). All continuous variables were standardised to avoid their larger measurement scales dominating correlation coefficients (Zuur et al. 2009). I utilised an information theoretic (IT) approach to construct eight candidate models (including the null model) that represented the combined ecological influences of rainfall, vegetation and reproductive success (Table 5.4). As the number of individuals captured the previous season and rainfall were likely to have an interactive effect on *D. blythi* occurrence an interaction was included in all relevant models. For similar reasons an interactive effect was also included in models with months since burn and vegetation classification. An interactive effect was not included between rainfall variables as I was interested in determining the influence of individual rain events rather than accumulated rainfall over time.

Table 5.4 Covariate structures of candidate models tested for *D. blythi* occurrence at Newhaven. * indicates interactive effect between dependent variables

<table>
<thead>
<tr>
<th>Model</th>
<th>Covariate Structure</th>
</tr>
</thead>
<tbody>
<tr>
<td>Null</td>
<td>~ 1</td>
</tr>
<tr>
<td>Short term</td>
<td>~ Rainfall 1 month prior * Ave. temperature 1 month Prior</td>
</tr>
<tr>
<td>Breeding</td>
<td>~ Rainfall previous summer * # individuals captured previous season</td>
</tr>
<tr>
<td>Young</td>
<td>~ Rainfall previous winter * # individuals captured previous season</td>
</tr>
<tr>
<td>Growth</td>
<td>~ Rainfall current summer * # individuals captured previous season</td>
</tr>
<tr>
<td>ReproSuccess</td>
<td>~ Rainfall previous summer + Rainfall previous winter + Rainfall current summer</td>
</tr>
<tr>
<td>Vegetation</td>
<td>~ Vegetation classifications</td>
</tr>
<tr>
<td>Structure</td>
<td>~ Vegetation classification * Months since burn</td>
</tr>
</tbody>
</table>
Models were ranked using the Akaike Information Criterion (AIC) with final model selection based upon four criteria; low AIC, low AIC difference (ΔAIC; difference between the AIC of the model with lowest AIC score and the particular model in question), high Akaike weight (\(w_i\); relative support of a particular model based on the models tested) and the amount of null deviance explained by the model (R\(^2\)\text{cond}); (Burnham & Anderson 2002; Zuur et al. 2009). These four criteria were used as they provide information regarding model variance as well as overall model fit. All top models were assessed for over dispersion using the Pearson dispersion statistic and patterns in residual to ensure model specification was appropriate. All modelling was conducted in lme4 package (Bates et al. 2014) in the statistical program R (R Core Team 2015).

5.3. Results

Across survey years, annual rainfall ranged from 110.7 mm to 568.8 mm at Newhaven and 77.4 mm to 590.6 mm at UKJNP. A total of 171 sites were sampled in Newhaven with approximately 58% detecting \(D. \ blythi\) at least once. At UKJNP a total of 124 sites were sampled with approximately 67% of sites successfully detecting \(D. \ blythi\) at least once.

Model results indicated \(D. \ blythi\) detection was relatively stable when the previous years annual rainfall was below 400 mm. Although a slight increase in detection was noted when rainfall fell below 250 mm, the variation in responses between sites was large. By comparison, when rainfall exceeded 400 mm a strong increase in \(D. \ blythi\) detection was apparent (Figure 5.2). Overall detection rates were higher at UKJNP than at Newhaven.
Based upon the results above only years with less than 400 mm rainfall were included in further analysis of *D. blythi* occurrence during periods of low rainfall. This resulted in the years 2000, 2001, 2002, 2006 and 2011 being removed from the UKJNP data and 2011 and 2012 from the Newhaven data. Grouping of sites within 1 km resulted in 19 g-sites at UKJNP and 85 g-sites at Newhaven being included in all further analyses.

From the data collected at the UKJNP, *D. blythi* was more likely to be detected in the central borefields area with no evidence that animals persisted in the east and west areas of the park during periods of low rainfall (Figure 5.3). Nevertheless, *D. blythi* was not detected every year at all sites in the central borefields area during low rainfall years.
Figure 5.3 Frequency of *D. blythi* detection at sites located within prime habitat during periods of low rainfall at UKTNP. Colour shading indicates vegetation; red = *Triodia basedowii* sand plains, orange = *T. pungens* sand plains, pink = dunes. Symbol size indicates the number of years sampled; larger = more years. Symbol colour indicates the proportion of years surveyed that *D. blythi* were detected; light grey = 0 – 40%; dark grey = 40 – 80%; black = 80 – 100%. Open circles indicate g-sites where *D. blythi* was not detected.

In contrast to UKJNP, the area where *D. blythi* was detected in low rainfall years at Newhaven was less consistent (Figure 5.4). With the exception of the north-west corner of the property, *D. blythi* was detected at least once in low rainfall years across most areas. The sites where *D. blythi* was detected on over 80% of surveys in low rainfall years did show some consistency; the majority occurring in the central band of ‘soft’ spinifex habitat that runs across the property. The two exceptions to this occurred in a band of ‘hard’ spinifex sand plains in the south of the property.
Figure 5.4 Frequency of *D. blythi* detection at sites located within prime habitat during periods of low rainfall at Newhaven Wildlife Sanctuary. Colour shading indicates vegetation; red = *Triodia basedowii* sand plains, orange = *T. pungens* sand plains, pink = dunes, blue = ephemeral lakes. Symbol size indicates the number of years sampled; larger = more years. Symbol colour indicates the proportion of years surveyed that *D. blythi* were detected; light grey = 0 – 40%; dark grey = 40 – 80%; black = 80 – 100%. Open circles indicate g-sites that never detected *D. blythi*.

Of the candidate models used to examine the relationship between the timing of rainfall and reproduction and growth periods, the model that included rainfall across all reproductive periods was the best at predicting *D. blythi* occurrence at Newhaven (Table 5.5). Notably, a relatively similar amount of deviance was explained by all the models tested, including the null model ($R^2$ cond = 0.53). This indicates that although the above variables were important in improving the predictive capabilities of the model a large proportion of the variance in *D. blythi* detection was explained by different sites and effort.
Table 5.5 Generalised linear mixed effect model outcomes for all candidate models tested. Detection of *D. blythi* at Newhaven Sanctuary was modelled with binomial distribution and a logit link function. Site was included as a random variable and log (number of trap nights) as a covariate. Df = degrees of freedom, AIC = Akaike’s Information Criterion, ΔAIC = AIC difference from top model, *wi* = Akaike weight and $R^2_{\text{cond}}$ = proportion of deviance explained in model (Nakagawa & Schielzeth 2013).

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>AIC</th>
<th>ΔAIC</th>
<th><em>wi</em></th>
<th>$R^2_{\text{cond}}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Repro-Success</td>
<td>6</td>
<td>492.6</td>
<td>0.00</td>
<td>0.95</td>
<td>0.61</td>
</tr>
<tr>
<td>Vegetation</td>
<td>5</td>
<td>499.1</td>
<td>6.55</td>
<td>0.04</td>
<td>0.51</td>
</tr>
<tr>
<td>Structure</td>
<td>8</td>
<td>501.9</td>
<td>9.38</td>
<td>0.01</td>
<td>0.53</td>
</tr>
<tr>
<td>Breeding</td>
<td>6</td>
<td>503.0</td>
<td>10.45</td>
<td>0.01</td>
<td>0.56</td>
</tr>
<tr>
<td>Young</td>
<td>6</td>
<td>505.7</td>
<td>13.10</td>
<td>&lt; 0.01</td>
<td>0.55</td>
</tr>
<tr>
<td>Growth</td>
<td>6</td>
<td>507.0</td>
<td>14.40</td>
<td>&lt; 0.01</td>
<td>0.53</td>
</tr>
<tr>
<td>Short Term</td>
<td>6</td>
<td>510.7</td>
<td>18.16</td>
<td>&lt; 0.01</td>
<td>0.56</td>
</tr>
<tr>
<td>Null</td>
<td>3</td>
<td>513.5</td>
<td>20.94</td>
<td>&lt; 0.01</td>
<td>0.53</td>
</tr>
</tbody>
</table>

On closer examination, of the top model only two variables were considered good predictors; previous winter rainfall and previous summer rainfall (Table 5.6). The deviance explained by the amount of rainfall the previous winter was considerably larger than that explained by the amount of rainfall the previous summer indicating that the former was the best predictor overall. Model results showed *D. blythi* was more likely to be detected when the previous winter rainfall was greater and when previous summer rainfall was lower.
Table 5.6 Generalised linear mixed model (GLMM) estimates for the top model for predicting *D. blythi* occurrence. Detection of *D. blythi* was model with binomial distribution and a logit link function. Site was included as a random variable and log (number of trap nights) as a covariate. Bold indicates significant variables; β = covariate coefficients, SE = Standard error and R² cond = amount of null deviance explained in variable.

<table>
<thead>
<tr>
<th>Variable</th>
<th>β</th>
<th>SE</th>
<th>R² cond</th>
<th>P value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Current summer rainfall</td>
<td>0.22</td>
<td>0.14</td>
<td>0.10</td>
<td>0.11</td>
</tr>
<tr>
<td>Previous winter rainfall</td>
<td>0.46</td>
<td>0.14</td>
<td>13.64</td>
<td>0.001</td>
</tr>
<tr>
<td>Previous summer rainfall</td>
<td>-0.63</td>
<td>0.15</td>
<td>4.04</td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>

5.4. Discussion

Overall, my results provide varying support for my initial predictions. As predicted, I found *D. blythi* showed an increased level of detection when rainfall increased above 400 mm the previous year. My predictions regarding the spatial contraction of *D. blythi* populations to key areas in low rainfall years was also supported in varying degrees at both sites although more strongly at Uluru Kata-Tjuta National Park. My results did not support my prediction that fire and rainfall post weaning would promote survival. Instead, the best predictor of *D. blythi* occurrence was greater rainfall prior to weaning.

Overall, *D. blythi* was detected more readily at Uluru Kata–Tjuta National Park than at Newhaven Wildlife Sanctuary. This is expected as the trapping protocol conducted at UKJNP was designed to target areas of *D. blythi* abundance whereas at Newhaven surveys were focused on more general monitoring across the broader landscape. Examination of data from both sites indicated a strong positive increase in *D. blythi* detection across a broader area when rainfall exceeded 400 mm which is likely due to greater productivity increasing resource availability and reproductive success (Letnic & Dickman 2006; Letnic et al. 2005). My findings concur with several previous studies on *D.*
*blythi* which show increases in population sizes and/or detection during periods of higher rainfall (Dickman et al. 2001; Gibson & Cole 1992; Greenville et al. 2016). However, a lack of response to rainfall by *D. blythi* has also been noted in several other studies also conducted in similar areas (Masters 1998; Masters & Dickman 2012). In these studies, the high annual rainfall totals only reached 302 mm at some sites and therefore may not have reached the 400 mm threshold required to stimulate a response. Furthermore, these studies had limited replication of sites in each location (n = 2) as they were not focused on detecting changes at a broad scale but instead were focused on the abundance of individuals within sites (Masters 1998; Masters & Dickman 2012). Locations of sites were situated in areas known to support stable *D. blythi* populations and therefore were likely to represent areas of refuge, which are capable of sustaining populations regardless of rainfall. Due to limitations within this study, I was unable to assess population-specific responses. To ascertain whether the response found here is indicative of populations more broadly, long-term monitoring across several populations, both in and out of refuge sites, is essential.

The stabilisation of *D. blythi* detection even at relatively low levels of rainfall supports previous work that low density *D. blythi* populations persist in these broad locations during periods of lower rainfall (Baker 1996). By examining the spatial patterns of detection during low rainfall years, I was further able to identify areas in the broader landscape where *D. blythi* are more likely to persist. In the Uluru Kata –Tjuṯa National Park, patterns of repeated occurrence were strong in the central borefields area. This finding suggests that this area is a refuge for the species. In contrast, although occurrence was consistently identified in the central regions of Newhaven *D. blythi* continued to
be detected sporadically across the property. The varying responses by *D. blythi* between the two locations may reflect different overall environmental conditions between sites. In general, Newhaven records greater mean annual rainfall (334.2 mm; BOM 2015b) than UKJNP (274.6 mm; BOM 2015c) and regularly experiences ‘top up’ rainfalls during the winter months, even during El Niño phases (BOM 2015b). Therefore, populations in UKJNP will likely experience more extreme conditions in El Niño phases, which results in a more dramatic spatial decline into refuge areas.

The areas of consistent use in each location are considerably different in vegetation structure (See Chapter 2 for details). At Newhaven, the area focused on the *T. pungens* sand plains whilst at UKJNP the area was dominated by *T. basedowii*. Generally, Newhaven experiences more regular wild fires than UKJNP (Commonwealth of Australia 2016; Turner et al. 2008) and therefore *D. blythi* may potentially select areas which show quicker regenerating capabilities at Newhaven, such as the dominant species *T. pungens* (Latz 2007). Given fire has been a long-standing component of Australian arid systems it is plausible that *D. blythi* populations are capable of varying their habitat selection depending on local fire histories. This would aid in reducing the influence of fire at a local scale whilst increasing their ability to persist across a broader landscape (Körtner et al. 2007; Thompson & Thompson 2007a).

At a local scale (within Newhaven), I found no support for the influence of fire when examining the drivers of *D. blythi* detection during periods of low rainfall. The only good predictors found in this study were rainfall during the previous summer and winter. Overall, the strongest response found was the increase in occurrence when higher rainfall was recorded during the previous winter. The winter period corresponds with the time young are being maternally cared for
prior to weaning (See Chapter 6) and is typically the time of the year with the lowest recorded rainfall (BOM 2015b). It is likely that greater rainfall during these periods would increase the availability of food resources, such as invertebrates (See Chapter 3; Kwok et al. 2016). Greater resources at this time would not only promote survivorship prior to weaning but also increase the body condition of young *D. blythi* at weaning and potentially increase survivorship during dispersal. Although a negative effect was found between *D. blythi* detection and increased rainfall the previous summer; the effect was weak (explaining less the 5% of the deviance in the model) and therefore should be considered with caution.

5.4.1. **Management considerations**

My study has provided important information that will aid in the management of *D. blythi* populations. I clearly identified a long-term response by *D. blythi* to variations in rainfall as well as more specific responses in individual populations.

The variations in responses found between locations highlight the importance of population specific management plans that identify the key areas within prime habitats. The factors influencing these responses may be of greater importance in the more southern populations of *D. blythi* as they show a more dramatic contraction in low rainfall periods. The potential reliance of *D. blythi* on a relatively small area of the landscape increases their vulnerability to changes in the habitat caused by fire and/or greater predation pressure. The interaction between fire and introduced predators has previously been identified as an important factor influencing *D. blythi* persistence. Individuals are known to be excluded or removed from areas with reduced cover during periods when introduced predator numbers are highest (Greenville et al. 2016). This is of
particular concern at UKJNP, which may experience potentially greater exposure to predators after fire due to the long regeneration time (~25 years) of dominant Triodia species (*T. basedowii*) in the area. However, as with refuge areas elsewhere, it provides a more clearly defined area of land in which to focus management efforts (e.g. Pavey et al. 2014).

I recommend both populations continue to be monitored regularly particularly after high winter rainfall as greater detection during these years may provide more substantial data that can be further utilised to assess broader population fluctuations during periods of low rainfall.
CHAPTER 6

HABITAT USE OF THE BRUSH-TAILED MULGARA

(*DASYCERCUS BLYTHI*)

Top: Field assistant recording movement pathways (J Molyneux)
Bottom: D. blythi released with spool attached (J Molyneux).
Abstract
One of the few small to medium-sized Australian mammals that has persisted across its distribution despite increasing pressures is the brush-tailed mulgara (*Dasycercus blythi*). Although *D. blythi* populations have declined since European settlement, they are currently considered stable. Small, low density isolated populations currently persist across the landscape during periods of low rainfall. The main threats to the species are currently thought to be the presence of introduced predators such as cats and foxes and changes in fire regimes. My study examined how the species utilises its surroundings to improve access to food whilst reducing exposure to predators during a post boom period. Through spool and line tracking, I examined the use of different habitat components by *D. blythi* and the subsequent exposure to predators. I found that all *D. blythi* in this study actively selected the open microhabitats between vegetation. Previous studies in similar habitats have shown open areas between *Triodia* not only support larger invertebrate prey but are frequently used by other small mammals and therefore may provide greater access to food. I found individuals experiencing greater physiological demands consistently used resource rich patches (such as termite mounds and shrubs) which would further facilitate more efficient and successful foraging and reproductive success. I also identified the repeated use of great desert skink burrows, which may provide access to prey items (such as young skinks), protection from predators and/or thermoregulation benefits.
6.1. **Introduction**

Knowledge of habitat requirements and/or the preference for particular areas are key to effectively managing and protecting species (Morris 2003; Stamps & Swaisgood 2007). The mechanisms driving habitat selection are often complex as individuals trade-off between competing factors such as food availability, predator exposure and competition (Stephens & Krebs 1986). The importance of selecting the best habitat for optimal foraging success is essential, particularly in environments where resources can sometimes be scarce and difficult to obtain, such as Australia’s arid zone. Australia’s arid zone is not only well known for its unpredictable climate and limited resources (Letnic et al. 2005; Letnic et al. 2013) but has also been subjected to one of the worst rates of extinction in recent times (Johnson 2006). Understanding how species utilise their surroundings to optimise access to resources is essential to ensure we support the persistence of remaining species in this region.

Australia’s arid zone is dominated by hummock grassland habitat, that covers approximately 22% of the continent (Griffin 1984). Australia hummock grassland is named and identified by its most dominant plant genus, *Triodia* (also known as spinifex), which grows to form hummocks up to one metre in height (Cofinas & Creighton 2001). These areas of hummock grasslands are further characterised by the sparse occurrence of over-storey species and the seasonal emergence of short lived ephemeral plants after rainfall (Cofinas & Creighton 2001). A main driver of cover in *Triodia* grassland habitats is fire, which occurs predominantly after several years of heavy rains. Large rainfall events cause an accumulation of fuel loads resulting in extensive fires, often covering many thousands of kilometres (Greenville et al. 2009). The historical occurrence of fire is well documented in the region (Greenville et al. 2009;
Griffin & Friedel 1985) with many species resilient to its occurrence through behavioural adaptations (e.g. Parr & Andersen 2006; Wright & Clarke 2008). Despite this, these fires are often considered important drivers of decline in species not only through direct mortality but also indirectly, through resource removal and increased exposure to predators, particularly feral cats and foxes (Kelly et al. 2012; Körtner et al. 2007; Masters 1993, 1996; Pastro et al. 2011).

Historically, species declines and extinctions in Australia’s arid zone have been greatest for critical weight range mammals (35 - 5500g; Johnson 2006; Woinarski et al. 2007). Despite this, the dasyurid marsupial *Dasycercus blythi* still survives in Australia’s *Triodia* grasslands and is one of the largest remaining native predators (50 - 120g; Van Dyck et al. 2008). It is known to persist in areas with limited cover, although are considered more vulnerable to predation by feral cats in these areas (Körtner et al. 2007; Masters & Dickman 2012). Compared to many other small mammals in the region *D. blythi* occupy relatively stable home ranges and are considered generalist feeders, consuming a variety of insects, other arthropods and small vertebrates (Chen et al. 1998; Masters 1998). Individuals can store fat in their tails and enter daily torpor to reduce energetic requirements, which further aids them when resources are scarce (Chen et al. 1998; Körtner et al. 2008; Masters 1998).

Although invertebrates are their predominant prey choice, *D. blythi* are also known to consume other small mammals, reptiles and birds (Chen et al. 1998; Masters 1998). Certain invertebrate taxa such as termites, which have high-energy gain, have been proposed as important component of their diet, particularly when other resources are depleted (Chen et al. 1998; Masters 1998; Mecke et al. 2013). Compared to other dasyurids in the region, *D. blythi* are known to consume a greater number of and larger prey due to their larger body
size (Chen et al. 1998). Although not yet investigated for *D. blythi*, other dasyurids often select open microhabitats as it increases access to the number (Bos & Carthew 2003; Friend et al. 1997; Haythornthwaite & Dickman 2006b) and size (Dickman 1988; Fisher & Dickman 1993a; Haythornthwaite 2005) of invertebrates.

Recent reclassification of *D. blythi* (Woolley 2005) has created ambiguity in regards to its historical distribution. However, the species is currently thought to occur in stable, scattered populations (Gibson & Cole 1992; Masters 1993; Masters 1998; Masters & Dickman 2012). Recently, it has been proposed that its successful persistence, compared to many other similar species in the region, is a result of a varied diet, ability to enter torpor and ability to withstand large fluctuations in body weight (Masters & Dickman 2012).

This study aims to identify how *D. blythi* utilises habitat components in the *Triodia* grasslands to optimise its foraging capabilities, and enable it to persist when other similar species have become extinct. I aim to identify the level of cover and the vegetation and substrate characteristics preferred by *D. blythi* as well as identify habitat patches that are used more intensively. Overall, I predict that:

1. *D. blythi* will utilise open areas to increase their own foraging capabilities, whilst maintaining proximity to cover to increase protection against known introduced predators, such as cats and foxes (Körtner et al. 2007).

2. *D. blythi* will select areas that increase access to food resources such as termite mounds, leaf litter and burrows.
3. The use of resource rich patches will be more pronounced in individuals with greater physiological demands, such as larger males and lactating females.

6.2. Methods

6.2.1. Study site and species
Newhaven Wildlife Sanctuary is situated 363 km north-west of Alice Springs, located on the eastern boundary of the Great Sandy Desert Bioregion. The area is considered semi-arid with a mean rainfall of around 300 mm.

The landscape is predominantly spinifex sand plains habitat, which was the focal habitat of this study. The sand plains are dominated by *Triodia pungens*, with intermittent, low-density pockets of low shrubs. The proportion of *Triodia* cover varies widely, from recently burnt areas (~15% average cover) to long unburnt areas (~45% average cover; see Chapter 3 for details). Common shrubs in the area include *Melaleuca glomerata* and *Grevillea juncifolia* with forbs and other small grasses prevalent immediately following rainfall in recently burnt areas.

This study was conducted in 2014 after 2 years of below average rainfall (188mm in 2012 and 271mm in 2013; BOM 2015b). During 2014, substantial rainfall in January and April yielded an annual total rainfall of 328.1 mm (see Chapter 2 for details).

A stable population of *D. blythi* has been recorded across the central regions of the sanctuary for the last five years. A large increase in their number and distribution seen in 2011 during a period of high rainfall (Chapter 5; AWC 2016a).
6.2.2. **Trapping protocol**

Quarterly trapping surveys were conducted over five surveys periods between November 2013 and November 2014 in the central spinifex sand plains habitat. At the beginning of each survey period, tracking surveys were conducted between three and five locations in the central spinifex sand plains habitat to determine areas where *D. blythi* sign was most abundant. On each occasion, the two sites with the greatest amount of sign were selected for trapping (Figure 6.1). A limit of two sites was selected to ensure sufficient trap effort was deployed at each site (between 675 and 1000 trap nights) and due to logistical constraints during field surveys.

![Figure 6.1](image.png)

Figure 6.1 (a) Location of successful (closed circles) and unsuccessful (open circles) *D. blythi* trap sites within the central spinifex sand plains within Newhaven Wildlife Sanctuary. (b) Insert map shows location of Newhaven Wildlife Sanctuary within Australia. Shaded areas show spinifex sand plains habitat with darker shading indicating the 2011-2013 burn areas and hatched areas signifying ephemeral lakes.

Each site was trapped over three or four nights using aluminium box traps (type A: 30 x 10 x 8 cm; Elliott Scientific Equipment, Upwey, VIC, Australia).

Depending on the number of traps available (which varied due to shared
resources with other research programs), box traps were laid in 7 - 11 parallel transects approximately 100 m apart and 500 m in length. Each transect consisted of 25 or 50 traps spaced at 20 or 10 metre intervals, respectively. Across all survey periods, nine sites were trapped with three resulting in *D. blythi* captures (Figure 6.1).

Traps were checked early morning, closed during the day and opened late afternoon. All bait was removed during the day and replaced each night to reduce ant activity around trap sites. In order to improve capture rates of *D. blythi* several bait types were tested over the survey period. Standard peanut butter/oats small mammal bait was used in all surveys and alternated down trap lines with a bait of sardines, dog kibble or cat kibble. For all *D. blythi* caught sex and breeding condition were recorded, basic measurements taken (pes length, tail length, tail width and scrotum width) and a passive transponder tag inserted subcutaneously in the interscapular region (Trovan Nanotransponder; Microchips Australia, Keysborough, VIC, Australia) to allow individual identification. All individuals were housed separately during the day in a large box (50 (l) x 40 (w) x 30 (h) cm) equipped with artificial and natural shelter, food and water. Animals were released, at their trap location, after sunset on the same day as capture.

### 6.2.3. Habitat use and availability

Data on habitat use were collected through spool and line tracking (Carthew 1994; Mendonca et al. 2010; Steinwald et al. 2006), as pilot studies showed this was the most reliable and effective way of assessing habitat use in *D. blythi* (Appendix 8). Spools comprised reverse-spun bobbins (approximately 300 - 350 m of 70 - 2 nylon quilting thread: Penguin Threads Pty Ltd, Prahran, Victoria, Australia) wrapped in black electrical tape and glued onto the fur of the lower
rump of the animal using cyanoacrylate ‘super glue’. Spools weighed approximately 4 g so, in line with ethical restrictions, only animals greater than 45 g were used. To reduce the chance of spools being shed as *D. blythi* moved through understorey vegetation and burrows, the electrical tape cover was tapered at the top end and then nestled in the fur ensuring no contact with skin (Figure 6.2).

![Image](image.png)

**Figure 6.2** Spool positioning on *D. blythi* rump

Animals were left alone immediately following release to reduce the influence of human presence on their behaviour. Within 48 hours of release all spool lines were followed, pathways recorded and thread removed from the site. To discount any initial flight behaviour caused by the presence of personnel during release, the initial 20 m of spool line or until the thread entered a burrow (which ever was shorter) was discounted. Subsequent inspection of the movement pathways indicated this period was sufficient to ensure normal activity had resumed. Spools were considered successful if a minimum of 30 m of non-flight activity was recorded.
Six habitat variables were recorded at either 1 or 4 m intervals along each spool trail (Table 6.1). Habitat variables were chosen to (1) identify broad scale movements in relation to fire scars, (2) identify how *D. blythi* moved in relation to different vegetation and substrate types identified in the area, (3) assess whether *D. blythi* selected areas with less cover and greater exposure to predators, and (4) identify areas of repeated use. A corresponding random trail was also assessed for each spool trail in order to quantify habitat availability. Random trails originated at the same starting point and were of equal length to their corresponding spool trail. Trajectory of random trails was determined through random bearings every 1 m. Individual *D. blythi* were only spooled once during each survey period.
<table>
<thead>
<tr>
<th>Variable</th>
<th>Interval recorded</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fire</td>
<td>1 m</td>
<td>Fire category with 1 m of sampling point; Unburnt (areas with no evidence of fire), Regenerating (areas with evidence of recent fire with regenerating vegetation), and Burnt (areas with evidence of recent fire without any regenerating vegetation).</td>
</tr>
<tr>
<td>Vegetation</td>
<td>1 m</td>
<td>Dominant vegetation type within 10 cm of sampling point; No vegetation, <em>Triodia</em>, <em>Triodia</em> edge, Shrub, Shrub edge, Underground or Grass edge</td>
</tr>
<tr>
<td>Substrate</td>
<td>1 m</td>
<td>Dominant substrate type within 10 cm of sampling point; Heavy leaf litter (&gt; 75% cover), Light leaf litter(&lt; 75% cover), Bare ground, Log, Termite mound, Underground</td>
</tr>
<tr>
<td>Cover – QCM</td>
<td>4 m</td>
<td>Estimate of % cover using the quadrant cover method (QCM); the area within a 1m radius of sampling point was divided into four quadrats using directional compass bearings. Habitat was assessed in each quadrat and from an aerial point of view to determine if features (vegetation, logs, etc.) would obscure the view of a predator standing 1 metre away (estimated at 30 cm height). Each quadrant and corresponding aerial view was given a score of zero (clearly visible), 10% (partially obscured) and 20% (fully obscured) resulting in a maximum 100% for each sampling point (Glen et al. 2010).</td>
</tr>
<tr>
<td>Distance to vegetation - DTV</td>
<td>4 m</td>
<td>Distance to closest vegetation; the area within a 1 m radius of sampling point was divided in four quadrats using directional compass bearings. Closest vegetation was measured within each quadrat to the nearest cm with a maximum of 100 cm recorded.</td>
</tr>
<tr>
<td>Bearing</td>
<td>1 m</td>
<td>Compass bearing to next 1 m interval marker (spool trials only).</td>
</tr>
</tbody>
</table>
6.2.4. **Statistical analysis**

6.2.4.1. **Habitat structure**

To estimate preferences for particular microhabitat variables, including vegetation and substrate types, the Manly-Chesson selectivity index (MCI; Manly et al. 1972) was calculated using the following formula:

\[
a_i = \frac{r_i}{p_i} \sum_{i=1}^{m} \frac{r_i}{p_i}
\]

where \(a_i\) is the selectivity index of a variable \(i\), \(r_i\) is amount variable \(i\) is used, \(p_i\) is the amount of variable \(i\) available and \(m\) is the number of variables in the environment. If \(a_i = \frac{1}{m}\), it means the microhabitat variable is used in proportion to its availability, whereas \(a_i > \frac{1}{m}\) indicates preference and \(a_i < \frac{1}{m}\) indicates avoidance. As \(m\) varied between individual spooling areas, the MCI was standardized by calculating the variation from \(\frac{1}{m}\), with highly negative results indicating avoidance and highly positive indicating preference. MCI was chosen as it allows for individual assessment of categorical habitat types and is suitable for the paired use and availability data collected for each individual in this study (Manly et al. 2002).

To further assess the selection of resources by individuals, I used the eigen analysis of selection ratios, as proposed by Calenge and Dufour (2006). This method maximizes the difference between use and availability on one or two factorial scores in order to assess variation between individuals. It is considered most suitable for categorical habitat types and design III type studies where use and availability are assessed for each individual. The eigenvectors were graphed to allow visual comparison of habitat selection by *D. blythi* groups (reproductive state, weight and sex) and habitat/survey attributes. Clustering of vectors in
groups indicated a consistent preference by the group. Analyses were conducted in the adehabitatHS package (Calenge 2013) in the statistical program R (R Core Team 2015).

As burrow availability and suitability was not adequately sampled in the availability data, all underground data points and their corresponding availability points were removed from analyses of microhabitat preference.

To assess habitat selection in respect to exposure to predators I calculated the average QCM (quadrat cover method) value, average minimum DTV (distance to vegetation) and average maximum DTV for each spool and random trail. Data from each D. blythi spool and matching random trail was compared using paired t-tests. Tests were initially run using all data points; however, as I was interested in exposure levels overall as well as outside potential shelter sites (burrows and Triodia), all analyses were conducted with and without potential shelter points. To determine whether QCM values and DTV, as recorded by spool lines, varied as a function of breeding condition, weight and/or month of capture, one-way analysis of variance (ANOVA) with Tukey HSD post hoc tests were conducted.

6.2.4.2. Movement pathways
To determine fine scale tortuosity (variation to linear movement), changes in bearing were recorded at each 1 m interval. Due to the bounded nature of the data (between 0 - 180°) bearing change was then divided by 180 resulting in a score between 0 and 1 which indicated the degree of change from no change to a complete turn-around, respectively. I evaluated the effect of habitat variables (vegetation type, substrate type, QCM, minimum DTV, maximum DTV and average DTV) on the level of tortuosity detected at any given point using
generalised linear mixed effect models with a binomial distribution and logit link function (Zuur et al. 2013). Animal ID was included as a random variable to account for multiple observations within individual spooling events. All models were compared to the null model to ensure increased model fit. To assess the effect of individual vegetation and substrate types on tortuosity, a Tukey HSD multiple comparison post hoc test was conducted. As bearing changes could not be detected in a burrow, all underground points were removed. All analyses of habitat variables were conducted using the lme4 (Bates et al. 2014) and multcomp (Hothorn et al. 2008) package in the statistical program R (R Core Team 2015).

Overall movement pathways were mapped using ArcGIS 10.1 (ESRI 2012) and visually inspected to determine areas of intensive use, defined by over 10 m of spool line repeatedly turning and crossing over itself in a small area. I calculated the difference between the proportion of vegetation and substrate types in and out each area of intense use and compared this to the intrinsic variation expected by individuals (average variation found across all spool lines). All differences greater than expected are presented along with information on the presence and type of burrows (as defined by Moseby et al. 2009c) in the areas of intense use.

6.3. Results

Over all trapping surveys a total of 11 356 trap nights resulted in 35 captures of 25 individual D. blythi, including 10 males and 15 females. Twenty spooling events were attempted on one male and 12 females (seven females were caught on multiple surveys). The remaining 15 captures were deemed unsuitable for spooling due to within survey recaptures (n = 2) or individuals being below the
required weight limit (n = 13). Spooling was successful for twelve individuals with spool length ranging between 50 and 315 m (average = 205 m; Table 6.2). No *D. blythi* moved between fire categories during a spooling event. The majority of females trapped in August had pouch young present and all females caught in November had empty pouches but were lactating and likely to be feeding young left in the burrow.

Table 6.2 Sex, body mass, reproductive condition and fire category of successful spooling events. ^ PY present indicates pouch young present.

<table>
<thead>
<tr>
<th>Month</th>
<th>Sex</th>
<th>Weight group (g)</th>
<th>Breeding condition^</th>
<th>Fire category</th>
</tr>
</thead>
<tbody>
<tr>
<td>May</td>
<td>F</td>
<td>70 - 80</td>
<td>Parous</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Aug</td>
<td>F</td>
<td>60 - 70</td>
<td>PY present</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Aug</td>
<td>M</td>
<td>60 - 70</td>
<td>NA</td>
<td>Regenerating</td>
</tr>
<tr>
<td>Aug</td>
<td>F</td>
<td>60 - 70</td>
<td>Parous</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Aug</td>
<td>F</td>
<td>50 - 60</td>
<td>PY present</td>
<td>Regenerating</td>
</tr>
<tr>
<td>Aug</td>
<td>F</td>
<td>60 - 70</td>
<td>PY present</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Aug</td>
<td>F</td>
<td>50 - 60</td>
<td>Parous</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Nov</td>
<td>F</td>
<td>50 - 60</td>
<td>Lactating</td>
<td>Regenerating</td>
</tr>
<tr>
<td>Nov</td>
<td>F</td>
<td>50 - 60</td>
<td>Lactating</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Nov</td>
<td>F</td>
<td>60 - 70</td>
<td>Lactating</td>
<td>Unburnt</td>
</tr>
<tr>
<td>Nov</td>
<td>F</td>
<td>60 - 70</td>
<td>Lactating</td>
<td>Regenerating</td>
</tr>
</tbody>
</table>

6.3.1. *Habitat structure*

Selection indices for vegetation types varied considerably between individual *D. blythi* (Figure 6.3a). The only exception was a consistent avoidance by *D. blythi* of the dominant vegetation type, *Triodia*. Overall selection ratios for substrate types also varied considerably between individuals with no substrate being consistently selected for or against by all individuals in the study (Figure 6.3b).
Although not included in the above analysis (due to difficulties in sampling availability), burrows represented between 0% and 13% (mean = 5%) of locations recorded for individual *D. blythi*.

Figure 6.3 Average Manly-Chesson selection indices (±SD) for all (a) vegetation and (b) substrate types.
The eigen analysis of vegetation preference produced two factors that explained 89.45% of the variation in data (55.63% for the first axis and 33.82% for the second axis). All points were skewed away from *Triodia* further indicating all individuals *D. blythi* avoided the use of *Triodia* (Figure 6.4). No consistent selection was apparent within *D. blythi* groups based upon reproductive condition, weight, habitat and survey attributes. However, unlike all the females recorded, the single male spooled showed a strong preference for both shrub and shrub edge.

Figure 6.4 Duality diagram of eigen analysis of vegetation selection ratios of *D. blythi* at Newhaven Wildlife Sanctuary. Top figure shows vegetation loadings on first two factorial axes; bottom figure shows individual *D. blythi* loadings on first two factorial axes. Individuals identified exhibiting within variable clustering are identified by blue = males; and red = females.
The eigen analysis of substrate preferences produced two factors that explained 95.20% of the variation in data (86.79% for the first axis and 8.41% for the second axis). Clustering of vectors was observed for several *D. blythi* groups (Figure 6.5). The single male sampled used slightly different substrate types to that of the females. He displayed a slightly higher preference for areas covered in both heavy and light leaf litter. A preference for bare ground and termite mounds was found for all the lactating females which had termite mounds available in the area surveyed (all surveyed in November) whilst females with pouch young (all surveyed in Aug) showed preference for areas of bare ground.

Figure 6.5. Duality diagram of eigen analysis of substrate selection ratios of *D. blythi* at Newhaven Wildlife Sanctuary. Top figure shows vegetation loadings on first two factorial axes; bottom figure shows individual *D. blythi* loadings on first two factorial axes. Individuals identified exhibiting within variable clustering are identified by green = males; red = lactating females with termite mound available; purple = females with pouch young; and blue = other females.
Assessment of potential exposure across all locations, as measured by the QCM, showed *D. blythi* selected areas that had significantly less cover (t = 6.45, df = 11, p-value < 0.005), greater maximum distance to vegetation (t = -2.72, df = 11, p-value = 0.020) and relatively similar minimum distance to vegetation (t = -1.54, df = 11, p-value = 0.151) than was available in the habitat (Figure 6.6).

After possible shelter sites (burrows and *Triodia*) were removed, habitat selection was relative to availability for all variables (QCM: t = -1.21, df = 11, p-value = 0.250; Min: t = 0.55, df = 11, p-value = 0.596; Max: t = 1.88, df = 11, p-value = 0.09). No difference was found in amount of cover or distance to vegetation used between breeding conditions (QCM:F2,8 = 0.03, p = 0.97; max: F2,8 = 3.60, p = 0.08; min:F2,8 = 1.76, p = 0.23), weight groups (QCM:F2,9 = 0.93, p = 0.43; max: F2,8 = 3.60, p = 0.08; min:F2,8 = 1.76, p = 0.23), fire category (QCM:F1,10 = 2.63, p = 0.14; max: F1,10 = 0.30, p = 0.59; min:F1,10 = 0.03, p = 0.87) or month of capture (QCM:F2,9 = 0.68, p = 0.53; max: F2,9 = 0.07, p = 0.94; min:F2,9 = 0.97, p = 0.42).
Amongst vegetation types, the only discernible difference in tortuosity was a greater linear movement (reduced bearing change) in areas without vegetation compared to areas under *Triodia* (Tukey HSD: \( p = 0.008 \)). No variation in tortuosity was found between any substrate types (Figure 6.7). No relationship
was found between the change in bearings made by *D. blythi* and the minimum (\(p = 0.141\)), maximum (\(p = 0.190\)) and average distance to vegetation (\(p = 0.141\)).

![Box plots showing change in bearing score for different vegetation and substrate types.](image)

Figure 6.7 Comparison of change in bearing across (a) vegetation and (b) substrate types for all spooling events. Letters depict statistically different vegetation types as determined by Tukey HSD post hoc test of GLMM results.

All movement pathways were mapped and four individuals were identified (through visual assessment) as utilising specific areas intensively (See Appendix 9 for spool pathway maps). All four *D. blythi* that intensively used areas were
females and no consistency was found in survey month, weight group, breeding condition or fire category (Table 6.3). The only consistent difference found in the high use areas was the presence of burrow entrances, particularly those of *Liopholis kintorei* (Great Desert Skink).

Table 6.3 Habitat and *D. blythi* characteristics of areas of high use determined through spool and line tracking.

<table>
<thead>
<tr>
<th>Survey month</th>
<th>Weight group</th>
<th>Breeding condition</th>
<th>Fire Category</th>
<th>Vegetation</th>
<th>Substrate</th>
<th>Burrow entrances</th>
</tr>
</thead>
<tbody>
<tr>
<td>May</td>
<td>70-80</td>
<td>Parous</td>
<td>Unburnt</td>
<td>↑ Triodia use</td>
<td>↓ leaf litter use</td>
<td>2 x L. kintorei</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>↑ bare ground use</td>
<td></td>
</tr>
<tr>
<td>Aug</td>
<td>60-70</td>
<td>Parous</td>
<td>Unburnt</td>
<td>No variation</td>
<td>No Variation</td>
<td>1 x unknown species</td>
</tr>
<tr>
<td>Aug</td>
<td>50-60</td>
<td>PY pres.</td>
<td>Regen.</td>
<td>No variation</td>
<td>No Variation</td>
<td>2 x L. kintorei</td>
</tr>
<tr>
<td>Nov</td>
<td>60-70</td>
<td>Lactating</td>
<td>Unburnt</td>
<td>No variation</td>
<td>No Variation</td>
<td>7 x L. kintorei</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2 x D. blythi</td>
</tr>
</tbody>
</table>

6.4. Discussion

This study is the first known investigation into fine-scale habitat use by *Dasycercus blythi*. Overall, my results showed considerable support for my initial predictions.

On a fine spatial scale, the results supported predictions that *D. blythi* would prefer open microhabitats (although not necessarily bare ground). While researchers have previously captured *D. blythi* in open habitats (Dickman et al. 2011; Körtner et al. 2007; Masters et al. 2003; Pavey et al. 2011; Thompson & Thompson 2007a), this is the first study to show that even in habitat with lower levels of cover (regenerating from fire), *D. blythi* prefer to use open areas between vegetation for foraging. As the largest remaining native marsupial
predator in the area, the dietary requirements of *D. blythi* are greater than smaller co-habitants (Fisher & Dickman 1993a, 1993b), and therefore it is likely it selects microhabitats that optimise their foraging capabilities. Previous studies in similar habitats have shown open areas between *Triodia* support larger invertebrate prey (Fisher & Dickman 1993a) and are frequently used by other small mammals that occur in the diet of *D. blythi* (Haythornthwaite 2005; Haythornthwaite & Dickman 2000; Murray & Dickman 1994). Although I was not able to make specific behavioural observations during this study, it is likely that by selecting these areas *D. blythi* would maximize foraging opportunities, enabling them to support their larger body size (Fisher & Dickman 1993a).

Foraging in the open spaces between *Triodia* has risks as predators that hunt visually may detect and capture *D. blythi* more readily in these areas. This was evident in a previous study on *D. blythi* where substantially more predation by introduced predators occurred in recently burnt habitats compared to unburnt habitat (Körtner et al. 2007). The greater risk-taking behaviour by *D. blythi* in the present study may be due to the changes in resource availability during a post boom period. As resources become scarce, *D. blythi* may display greater risk-taking behaviour in order to obtain sufficient resources (Luttbeg & Sih 2010), or alternatively as predator numbers decline post boom *D. blythi* may perceive lower levels of risk and therefore be content to increase risk-taking behaviours (Dickman et al. 2011). *D. blythi*’s preference for areas where predation risks are higher is not completely surprising. Native predators could include *Tyto alba* (Barn owl), *Ninox novaeseelandiae* (Southern boobook), *Elanus axillaris* (Black-shouldered kite), *E. scriptus* (Letter-winged kite), *Aspidites ramsayi* (Woma python), *Antaresia stimsoni* (Stimson’s python), *Pseudonaja nuchalis* (Western brown snake), *Pseudonaja textilis* (Eastern brown snake),
Pseudechis australis (Mulga snake) and Varanus giganteus (Perentie). Although T. alba, N. novaeseelandiae, E. axillaris, V. giganteus and A. ramsayi are common in the region and capable of preying upon D. blythi; however, they are known to prefer smaller prey and/or reptiles and therefore would not likely create substantial predation pressure when preferred prey is available (Debus et al. 2006; Dickman et al. 1991; Penck & Queale 2002; Shine & Slip 1990). E. scriptus is a crepuscular hunter that generally occurs only in large numbers when rodent numbers are high during boom periods, therefore predation pressure from this species would also be restricted (Pavey et al. 2008). Although more open areas are considered higher risk in relation to predation by introduced predators, many native species, particularly snakes, forage in more closed habitats (Burton 2013; Read et al. 2011). By selecting open areas between Triodia, D. blythi may avoid predation by native predators whilst increasing access to prey and the ability to visually detect predators, such as cats that also rely upon vision to stalk smaller prey (Dickman 1996b).

Although D. blythi selected areas with higher exposure overall, by removing shelters sites (underground and Triodia), I was able to determine this was due to the above avoidance of Triodia rather than an overall selection of more exposed areas. By maintaining proximity to cover D. blythi would likely increase their ability to avoid predators as it would facilitate a quick retreat to shelter sites, such as burrows or the needle-leafed Triodia (which is likely to provide effective protection from larger predators; Dickman et al. 2011). This type of behaviour has been seen in several similar species in the wild (Haythornthwaite 2005; Haythornthwaite & Dickman 2000; Kotler 1984) and in both wild and captive D. blythi populations (Woolley 1990; J Molyneux 2006, observation). My results also showed D. blythi moved in a more linear fashion when utilising
areas without cover. Greater linear movement has previously been thought to reduce predation risk as individuals can move at greater speed and therefore spend less time in those areas (Jacob & Brown 2000; Kotler et al. 1991; Stokes et al. 2004). The level of tortuosity recorded by spools in open areas is likely to be underestimated as lower levels of vegetation would reduce the chance of the spool line catching and therefore representing the true pathway of the individual. More detailed behavioural studies on *D. blythi* would provide insight into whether my results are caused by limitations of the technique used here or a strategy to reduce predation risk whilst maintaining access to areas where resources are higher.

The individual variability in selection of all other habitat components was similar to results on other arid zone dasyurids (Bleicher & Dickman 2016) and may be indicative of individual behavioural strategies. As *D. blythi* are considered sedentary (Gibson & Cole 1992; Masters 1998; Masters & Dickman 2012) and persist in areas, even during periods of low rainfall (See Chapter 5; Masters & Dickman 2012), developing individual strategies may reduce intra-specific competition when resources are scarce (Bleicher & Dickman 2016). Although my study showed further support for this in the individual selection of resource rich patches (discussed below), further studies with larger sample sizes across periods of varying resource availability would help clarify this.

This study partially supported my second prediction that *D. blythi* would prefer resource rich habitat patches. Although sample sizes were small, examination of areas of concentrated use did show an intense level of activity around burrows, particularly those of *Liopholis kintorei* (great desert skink). Previous studies have found other dasyurid species selectively utilise areas where burrows of prey species
are present, likely increasing foraging success (Haythornthwaite 2005). Although previous *D. blythi* diet studies did not classify prey items to species levels, skinks are considered important components of *D. blythi* diets (Chen et al. 1998; Masters 1998). As *L. kintorei* young take several years to reach full size (McAlpin 2001) they would provide an important food item suitable for *D. blythi* throughout the year. Actively hunting young *L. kintorei* would provide substantial benefit to *D. blythi* as they occur in predictable locations (burrows) (McAlpin 2001), would provide substantial energetic gains (Chapman & Reiss 1999) and may also be easy to handle in cooler weather when activity is low (McAlpin 2001). Recent evidence also shows *L. kintorei* burrows are utilised by a variety of other fauna species and could be a potential resource hotspot for generalist predators like *D. blythi* (Moore et al. 2015). I propose that burrows, particularly *L. kintorei* burrows, provide important food resources for *D. blythi* at Newhaven. The greater use of burrows may also reflect a level of predator awareness by *D. blythi*. The use of burrows to protect against introduced predators has previously been proposed as a key behaviour for the survival of *D. blythi* above other species that have experienced dramatic declines in the region (Masters & Dickman 2012). The use of deep burrows is also thought to help in thermoregulation and protect against the extreme conditions found on the surface (Woolley 1990). Although both these explanations are sound, the convoluted movement paths of *D. blythi* in and around burrows in this study indicate movement more representative of foraging. Further research into the connection between the use of *L. kintorei* burrows and *D. blythi* diet during these periods would provide insight into the importance of the relationship between these two species.
My third prediction regarding the more prolific use of resource rich patches by individuals with greater physiological needs was partially supported. The single male spooled showed a clear preference for areas under shrubs, shrub edges and areas where leaf litter was present. Findings from previous studies in similar habitats (Bos & Carthew 2003) showed that invertebrate abundance and foraging opportunities are high in these areas. Although the male sampled here would not be considered particularly large (weight of 68.5 grams), the spooling was conducted in August when males are known to undergo a severe decline in body condition (Gibson & Cole 1992). When compared with previously recorded condition for male D. blythi post breeding (tail width = 5.2 - 6.5 mm; Masters & Dickman 2012), the male sampled in this study was considered relatively healthy (tail width = 8 mm). Greater shrub cover has previously been thought to help sustain individuals during periods of lower resources (Dickman et al. 2011) and may have aided the male sampled here in maintaining better condition. Future sampling across multiple individuals is needed to clarify potential correlations between body condition and habitat selection.

Interestingly, the breeding condition of females also indicated some preferences for certain habitat characteristics. Open areas were consistently preferred by females carrying pouch young, whilst lactating females with young in burrows preferred termite mounds (if available in the area sampled) as well. As previously discussed, open areas between Triodia are one of the areas known to support larger food resources and termites are a known high energy food source readily eaten by D. blythi (Chen et al. 1998) and a variety of other vertebrate species (Abensperg-Traun & Boer 1992; Abensperg-Traun & Steven 1997). Both the early and later stages of raising young are not only considered energetically costly in dasyurids (Geiser & Masters 1994) but are also the only periods when
*D. blythi* are known not to utilise the energy saving behaviour of torpor (Geiser & Masters 1994; Körtner et al. 2008). In order to meet the energetic demands of carrying young and lactation it is not surprising that *D. blythi* with pouch young exploit open areas where resources are highest. Lactating females with young in burrows experience high energy requirements as well as being potentially limited in movement in order to protect defenceless young in burrows from predation (Bos & Carthew 2007a; Gibson & Cole 1992; Woolley 2005). By seeking out termite mounds lactating *D. blythi* would be able to obtain large amounts of food resources quickly, reducing the time spent foraging and increasing the time available to protect young in burrows.

My findings have considerably increased our knowledge on how *D. blythi* utilise their surroundings to optimise foraging during periods when resources are limited. Considering the potential isolation of populations during these periods this information will help to develop more precise, targeted management strategies to ensure the species continue to persist into the future.
CHAPTER 7

EFFICACY OF THREE POPULAR TECHNIQUES FOR DETECTING AND MONITORING ARID ZONE SPECIES IN A SPINIFEX SAND PLAINS HABITAT

Clockwise from left: Goanna tracks (J Molyneux), Box trap (J Molyneux), Pitfall and drift fence (P Moore), IR camera trap with scent lure (J Molyneux)
Abstract
Accurately monitoring and assessing biodiversity is a key component of achieving and implementing effective management strategies. Landowners in Australia’s arid zone face many unique challenges, particularly because they manage vast remote landscapes that support numerous cryptic species that are difficult to monitor. Although researchers and land managers are increasingly utilising a wider variety of monitoring methods to improve detection across species and areas little is known regarding their ability to achieve this. This study aims to assess the efficacy of three popular monitoring techniques utilised in Australia’s arid zone; live trapping, sign surveys and infra-red (IR) cameras. I explored variations in capture rates and species richness for each method and assessed initial and on-going costs of the methods over time. My results indicated greater capture success when cameras were deployed with lures and when more experienced personnel were utilised for sign surveys. Trap type was found to considerably alter capture rates within taxa, with small mammals more readily caught in box traps and reptiles in pitfalls. Overall, sign surveys detected the greatest number of species/groups. However, when focused on target species IR cameras detected more, although the probability of detection by each method varied between specific species. Although IR cameras were initially the most expensive method, the low ongoing costs of maintaining cameras in the field meant that they became the most cost effective after eight survey periods.
7.1. Introduction

Wildlife surveys have long been the most common method of collecting biodiversity information to successfully assess, manage and conserve environments (Baillie et al. 2004; Boitani & Fuller 2000; Southwood & Henderson 2000). One of the most fundamental factors influencing successful data collection is the selection of suitable survey methods (Garden et al. 2007; Thompson & Thompson 2007b). Several studies have shown survey methods to vary considerably in their effectiveness in detecting species, with no single method considered optimal across all species or habitats (Catling et al. 1997; Ellis 2013; Garden et al. 2007; Lyra-Jorge et al. 2008; Silveira et al. 2003; Southgate et al. 2005; Stokes 2013). Furthermore, these studies have also shown considerable differences in financial costs. With increasing pressure to conduct more cost effective and accurate surveys, selecting the optimal survey methods is paramount. In Australia’s arid zone little information is available on the effectiveness of different methods to accurately detect species in a cost effective manner.

Land managers in Australia’s arid zone face many challenges, notably the large spatial scales, remote locations and a unique and diverse range of species (Byrne et al. 2008; Letnic et al. 2011; Morton et al. 1995; Perry & Goodall 1979; Smith & McAllister 2008). Most studies in the region have relied on live trapping to detect species with cage, box, funnel and pitfall traps primarily used to sample small mammals and reptiles (e.g. Dickman et al. 2010; Greenville et al. 2013; Masters & Dickman 2012; Pavey et al. 2014). The benefit of live trapping is clear, as physical capture provides unambiguous and individual-specific information that is required to determine important population parameters (Jones et al. 1996; Nichols & Pollock 1983). However, live trapping has also been
criticised as financially costly, time consuming and ineffective for capturing some cryptic species (Stanley & Royle 2005; Wiewel et al. 2007). Physical capture also requires individuals to be detained for periods causing stress and possible disruption to natural activity patterns.

In contrast, sign surveys are a non-invasive method also regularly used for detecting animal species. They involve searching for evidence (i.e. track, scats, burrows, etc.) of a species in the landscape as a means to detect their presence. Due to suitability of sandy substrates and the distinctive nature of the sign left by many species, this method has been particularly useful for detecting rare and cryptic species in the arid zone (Bennison et al. 2014; Southgate et al. 2005). Sign surveys are usually considered very cost effective as no equipment is required and they can be conducted relatively quickly. This method is not without drawbacks as the ability to detect sign can be dramatically reduced due to varying environmental conditions, such as wind or rain, and the skill of the personnel, due to many species leaving similar tracks that require a high level of expertise to discriminate between (Moseby et al. 2009c). As they record only absence or presence sign surveys are also limited in their ability to provide accurate abundance estimates; however, continual advances in statistical methodologies have improved our ability to accurately utilise this data (e.g. He & Gaston 2000; Hui et al. 2009; Royle & Nichols 2003; Stanley & Royle 2005; Webb & Merrill 2012).

In recent years alternative non-invasive methods, such as infra-red (IR) cameras, have increased in use to complement or sometimes replace live trapping and sign surveys (Barea-Azcon et al. 2007; De Bondi et al. 2010; Meek et al. 2014; Silveira et al. 2003; Welbourne et al. 2015). De Bondi et al. (2010) found infrared cameras with lures were especially successful in detecting cryptic
species that have previously been considered difficult to monitor through live trapping regimes. The ability to deploy cameras quickly and across large scales has made it an increasingly appealing choice in many habitats and regions (e.g. Bengsen et al. 2011; Biebouw & Blumstein 2003; du Preez et al. 2014; Fairfax et al. 2012; Noss et al. 2012; Silveira et al. 2003). However, the high initial costs of purchasing cameras mean they are not always a viable option.

Across Australia’s arid zone all of the above methods have been proposed as suitable for application across a range of vertebrate species, and they are used by a variety of land managers (AWC 2016a; National Health and Medical Research Council 2013, 2015; Watson 2007). However, to date no study has directly compared the efficacy of all these methods and their ability to detect a range of arid zone species. In order to ensure that species detection data is accurate and suitable for the basis of regional management decisions, such comparative studies are essential.

This study is the first to directly compare three different survey methods of monitoring terrestrial vertebrate species; live trapping, sign surveys and infra-red camera trapping, in one of Australia largest habitats; spinifex sand plains. This study will examine how variations in deployment of methods can alter capture/detection success as well as compare the species richness detected by each method overall. I will focus in on several key species to assess the difference in the probability of detection between methods and compare overall financial costs over time. I predict that:

1. Cameras with lures and more experienced tracking personnel will capture and detect more species (both mammal and reptile) than cameras
without lures and novice personnel, respectively, whilst the trap type utilised by live trapping will target species-specific traits.

2. IR cameras will detect higher initial species richness than sign surveys and live trapping; however, across multiple surveys the number of species detected by live trapping will be similar to that of IR cameras.

3. The probability of detection for individual species will reflect species-specific traits that favour the detection by each method (e.g. highly identifiable tracks or attraction to bait).

4. Initially IR cameras will be have the greatest costs due to capital expenditure; however, due to low ongoing costs this will reduce over time to become comparative to both live trapping and sign surveys.

7.2. Methods

7.2.1. Study sites
The study was conducted across two locations in the Northern Territory; Newhaven Wildlife Sanctuary and Yulara borefields (See Chapter 2 for full description of study sites). Five and three sites were surveyed across Newhaven and the borefields, respectively (Figure 7.1). Following the design of previous studies in similar habitats (Dickman et al. 2011; Masters 1993, 1996) neighbouring sites were a minimum of 1 km apart.
Figure 7.1 Site locations for all surveys conducted at (a) Newhaven Wildlife Sanctuary and (b) Yulara borefields.

Each site was surveyed three times in 2014. The five sites in Newhaven were surveyed in February, May and August and the three borefields sites in April, July and October. During each survey period all survey methods were conducted across the same 2 ha grid at each site to ensure consistency of species occupancy/detection between methods (Figure 7.2).
7.2.2. **Live trapping**
During each survey period live trapping was conducted at all sites (for sampling protocols of small mammal and reptile live trapping see Chapter 3).

7.2.3. **Sign surveys**
During each survey, any animal sign detected (i.e. tracks, scats, burrows, diggings and animals) was recorded by trackers whilst traversing parallel transects in a 2 ha area surrounding each trap site (as per Moseby et al., 2009). Independent surveys were conducted simultaneously by observers with one of two levels of experience (novice and intermediate) for a total of 0.5 person hours each (i.e. one tracker for 30 minutes or two trackers for 15 minutes). A novice was defined as a person without any, or very little, experience in sign surveys. During the search novices were supplied with two books; Triggs (1996) and Moseby et al. (2009c), to aid in identification. New novice trackers were used each survey period to ensure consistency in experience levels over the course of
the study. A person was classified as an ‘intermediate tracker’ if they had a moderate level of experience/skills; moderate is defined as the ability to detect and identify most animal sign in the area without assistance. Although the same intermediate tracker was used throughout the study, I compared the level of identification by the intermediate tracker to that of a local expert tracker at the completion of the study, which confirmed they had not gained enough experience to progress to expert. All surveys were conducted either 2 - 4 hours after sunrise or 1 - 3 hours before sunset to ensure optimal environmental conditions (i.e. light). Sign was recorded each time it was seen to the most precise level possible with species level being ideal. A series of tracks or runways were counted as a single sign as were large burrows with multiple entrances (e.g. great desert skink or spinifex hopping-mice).

7.2.4. **Infra-red (IR) camera surveys**

At each site, four Reonyx HC500 semi-covert infrared cameras (REONYX, Inc, Holmen, Wisconsin, USA) were deployed continuously between each live trapping survey period, resulting in nine months of data. Based on methods from previous studies on small to medium sized mammals (Fancourt 2014; Paull et al. 2011) and the minimum distance recommended by Kays et al. (2009), cameras in each site were spaced approximately 50 m apart to ensure spatial independence. From the centre of the live trapping grid I placed the four cameras 35 m along transects following the four cardinal directions. Prior to this study a short pilot study was used to determine the optimal camera positioning and bait/lure selection to maximise capture rates and identification of small mammal and reptile (J Molyneux 2012, unpubl.). Cameras were attached to a picket 20 cm above the ground (measured from the base of the camera), on a downward angle of 20°. Two cameras on opposite cardinal directions had a scent
lure of cotton wool soaked in sardine oil (selected for longevity, to prevent removal by ants and to maintain high levels of interest from species). Lures were placed one metre in front of the camera in a perforated, lidded PVC pipe (3.5 cm diameter, 30 cm long) buried to a depth of 25 cm to stop animals removing or interfering with them. The remaining two cameras were deployed with no attractant. During the pilot study substantial camera failure due to high temperatures was apparent, therefore, all cameras were surrounded with an insulated reflective cover to protect from direct sunlight as well as radiant heat from the ground.

All cameras were programmed to take two pictures per trigger, with a ‘RapidFire’ picture interval and fast shutter night mode. At the beginning of each live trapping survey, lure scent baits were replaced, cameras were checked and pictures downloaded. All cameras recorded continuously between survey periods. All photos taken were individually inspected and any animal identified to the most precise level of classification possible. During photo inspection, delineation between separate photo events (a series of photos of the same animal that activates the camera multiple times) was also recorded. In the first 3 months of the study a burrow was excavated in front of a single camera at site 10 (Newhaven) resulting in an exceptionally high activity and number of photos being taken. All photos from this camera were discarded to prevent bias in the data set.

7.2.5. Statistical analysis

To standardise the data and account for variations in trap effort by each survey method I converted all data from IR cameras and live trapping to capture rates (number of captures/number of trap nights). Similarly, to standardise the data for comparison between survey methods I utilised the initial 21 days of camera
data from each site. This represented a similar number of trap nights (N = 84) to live trapping (N = 87). In order to standardise the sign surveys I utilised only the data from the intermediate tracker as this was considered a standard effort for an individual survey period.

To compare between survey methods I investigated differences in species richness detected over a single survey period (SSR; number of species detected in a single survey period) and across all survey periods (ASR; accumulated number of species detected across all three survey periods). For both SSR and ASR I examined species richness in four ways: (1) all species – all mammal and reptile species detected; (2) target species – reptile and mammal species that could be detected by all methods (i.e. large species discounted as they could not be sampled via pitfalls or box traps) and those identified to at least genus level; (3) mammals – all target mammal species; (4) reptiles – all target reptile species.

Capture rates by each method and species richness between methods were analysed using generalised linear mixed effect models (GLMMs). GLMMs were used as they accounted for the repeated spatio-temporal survey design by including site and survey as a random covariates. All models included the primary covariate of interest; survey method, along with location (Yulara borefields or Newhaven) to investigate location specific differences. Model specification followed the equation below.

$$\text{Species Richness}_{ij} = \beta_1 + \beta_2 \times \text{survey technique}_{ij} + \beta_3 \times \text{location}_{ij} + \alpha_i + \delta_i + \varepsilon_{ij}$$

where

$$\alpha_i \sim N(0, \sigma_{\text{Site}}^2)$$

$$\delta_i \sim N(0, \sigma_{\text{Survey}}^2)$$

$$\varepsilon_{ij} \sim N(0, \sigma^2)$$
To assess the significance between individual methods post-hoc Tukey’s HSD tests were performed on all models. All models were initially generated using a Poisson distribution with a log link function; however, due to over dispersion in models (pearson statistics > 1; Zuur et al. 2012), a negative binominal distribution with a log link function was used which accounted for the variance in the data by the addition of an extra parameter (k or theta; Zuur et al. 2009). All models and analyses were conducted using the packages glmmADMB (Skaug et al. 2014) and multcomp (Hothorn et al. 2008) in the statistical program R (R Core Team 2015).

We further compared between survey methods by examining the probability of detection (POD) of all suitable species. Species were considered suitable if there were sufficient data (recorded by a minimum of two methods on 10 occasions), high precision (species level classification) and could be captured by all methods (i.e. cat was not included as it could not be live trapped due to the small trap types used). POD’s were estimated through maximum likelihood based multi-season occupancy modelling (MacKenzie et al. 2003) which allowed for variation in detection caused by sampling in different months. Multi-season occupancy models incorporate four parameters; $\psi_i$, the probability a species initially occupies site $i$; $p_{ij}$, the probability a species is detected at site $i$ on survey $j$ given that it is present at the site; $\gamma_t$, the probability than an unoccupied site in season $t$ is occupied by the species in season $t + 1$; and $\epsilon_t$, the probability that a site occupied in season $t$ is unoccupied by the species in season $t + 1$. I modelled inter-seasonal variation so that occupancy at a site in season $t$ is dependent on occupancy in the season prior, $t - 1$ or $\psi_{t+1} = \psi_t(1 - \epsilon_t) + (1 - \psi_t)\gamma_t$. Further details on multi season occupancy models are given in MacKenzie et al. (2003).
We included a single covariate; survey method, to predict the detectability in all occupancy models as this was the only variable of interest for the purpose of this study. I compared each model with the corresponding null model and assessed model fit through examination of the AICc (suitable for small sample sizes) with the lowest AICc considered most parsimonious (Burnham & Anderson 2002). All models tested are presented along with PODs for models containing method as a covariate. All occupancy modelling was performed in the unmarked package (Fiske & Chandler 2011) in the statistical program R (R Core Team 2015).

Lastly, I compared the total survey costs of each method based upon the initial capital costs and ongoing field survey costs. Capital costs covered all equipment purchases such as traps for live trapping, identification books for sign surveys and IR cameras (incl. memory cards, batteries, mounting poles and lures). Ongoing survey costs included vehicle costs, accommodation, food, wages and consumables. I based all survey costs upon sampling eight sites across a single location (situated 400 km from an origin) which was comparative to the location and effort in other arid zone studies in Australia (e.g. Dickman et al. 1999; Fisher & Dickman 1993b; Masters 1998; Pavey et al. 2008). To estimate the length of surveys I included two days for travel and camp set up and pack down. Seven days were allocated for live trapping as a maximum of four sites could be sampled at any given time (considering the ethical requirements to clear sites before temperature and ants became an issue in central Australia) and a day was required to clean equipment. I allocated three days for sign surveys as they could only be conducted at specific times and when weather was favourable. A single day was allocated for camera surveys, as personnel were only required to replace batteries and memory cards. I added an additional two days to the first live trapping survey to set up pitfalls (we assumed pitfalls remained onsite
between subsequent surveys) and one additional day to the first camera survey to set up mounts. I also included an additional survey trip to IR camera surveys to collect equipment at the completion of the study.

Vehicle costs were estimated at approximately $130/day, including $90/day for commercial 4WD car hire (Thrifty Car Rentals 2016) and $40/day for fuel. An additional $150 of fuel was added to cover travel to and from the survey location (approximately 400 kms each way). I calculated food at $50 pp/day and accommodation at $15 pp/day, based on average camping rates in remote areas (AWC 2011; Department of National Parks Sport and Racing 2016; Halls Creek Travel & Tourism 2016). Consumables were calculated at $100 per survey and an additional 1% rate of replacement for all equipment (capital costs) per survey. The rate of replacement (4% per year) was considered ample based upon this and other studies (Welbourne et al. 2015). Due to the remote location a minimum of two personnel were required on all field surveys. To calculate wages I included one experienced personnel and one volunteer for live trapping and camera surveys and two experienced personnel for sign surveys to ensure sufficient experience in trackers. Experienced personnel were paid for 8 hours each day of the survey period and cost $50/hr which represented hourly rate for an experienced (post graduate qualified) casual research assistant (Charles Darwin University [CDU] 2014; Welbourne et al. 2015). I also included three hours of post survey data entry by a less experienced (entry level) casual research assistant (cost $30/hr; CDU 2014) for sign surveys and live trapping and 12 hours of photo identification by a high level casual research assistant for camera surveys. I projected the total cost of running each method over 12 survey periods to determine overall costs over time.
7.3. Results

Across all surveys I conducted a total of 2,664 trap nights of live trapping surveys, 24 hours of sign surveys and 2,880 trap nights of camera trapping resulting in 2,979 individual records of animals and sign for mammals and reptiles (Table 7.1, Appendix 10, Appendix 11).

Table 7.1 Number of reptile and mammal ‘captures’ for each survey and deployment technique across eight survey periods during 2014.

<table>
<thead>
<tr>
<th></th>
<th>Live Traps</th>
<th>Sign Surveys</th>
<th>Camera traps</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Box Pitfall</td>
<td>Novice Inter</td>
<td>Lure No Lure</td>
</tr>
<tr>
<td>Mammal</td>
<td>47 2</td>
<td>239 557</td>
<td>681 384</td>
</tr>
<tr>
<td>Reptile</td>
<td>6 48</td>
<td>180 706</td>
<td>100 29</td>
</tr>
<tr>
<td>Total</td>
<td>53 50</td>
<td>419 1,263</td>
<td>781 413</td>
</tr>
</tbody>
</table>

7.3.1. Capture rates by methods

Box traps identified six species of small mammal and yielded higher capture rates than pitfalls (HSD; \( p = 0.01 \)) which only caught a single species; *Dasycercus blythi* (Figure 7.3, Appendix 10). In contrast, pitfalls caught significantly more reptiles than box traps (HSD; \( p < 0.001 \)) with pitfalls identifying eight species and box traps only two species; *Ctenotus hanloni* and *C. grandis* (Appendix 11).

The only taxa and method that showed variation between locations was the capture rates of small mammals by IR camera, which was lower at the borefields (GLMM; \( x = -1.48, p = 0.002 \)). Although overall cameras with lures yielded significantly higher captures rates of small mammals (HSD; \( p = 0.01 \)) further investigations showed this trend was only apparent at the borefield sites.

Cameras with and without lures both detected the same small mammal species, with the exception of the group *P. hermannsburgensis/M. mus,* which was only
detected by cameras with lures. Reptile species were also detected by cameras with and without lures. *C. pantherinus* the only species to show a clear difference (> 25) in the number of photo events taken by cameras with (n = 58) and without lures (n = 5).

Intermediate trackers identified significantly more tracks of both mammals (HSD; p < 0.001) and reptiles (HSD; p < 0.001, Figure 7.3). Both novice and intermediate trackers identified a similar number of species/groups of mammals (11 and 12, respectively) and reptiles (nine and 11, respectively). The species/groups only identified by intermediate trackers were non-specific rodents, legless lizards and *Moloch horridus*. 
Figure 7.3 Capture rates of (a) mammal and (b) reptiles for each deployment method used in (i) live trapping, (ii) camera traps and (iii) sign surveys. Letters indicate statistically different groups based upon post-hoc Tukey’s HSD test of generalised linear mixed effects models.

7.3.2. **Species richness between methods**

Location (Newhaven and borefields) was not a significant variable in any of the models tested and is therefore not considered further. Comparison between survey methods indicated that in individual surveys (SSR) and across all surveys (ASR), sign surveys detected the greatest total species richness, followed by IR cameras and then live trapping (HSD; p < 0.003 for all pairings; Figure 7.4). For examination of target species 50% of the species/groups were removed due to identification or detection issues. One, eight and four taxa were removed from live trapping, sign surveys and camera surveys respectively, as they failed
to be identified to at least a genus level. An additional 10 and two taxa were removed from sign surveys and camera surveys, respectively, as they were unable to be detected by at least one other method. When non-target species were removed, IR cameras detected the greatest richness in individual surveys (SSR) (HSD; sign surveys, p = 0.048; live trapping, p < 0.001) and sign surveys detected similar species richness to live trapping (HSD; p = 0.263). In contrast, richness of target species across all surveys (ASR) did not vary significantly across any survey method (HSD; Sign/Camera: p = 0.073; Trapping/Camera: p = 0.326; Trapping/Sign: p = 0.826).

By examining individual fauna classes I can discern that the difference in the SSR detected between methods overall was due to a greater number of mammal species being detected (HSD; Sign/Camera: p = 0.034; Trapping/Camera: p < 0.001; Trapping/Sign: p = 0.047) rather than reptiles (HSD; Sign/Camera: p = 0.759; Trapping/Camera: p = 0.759; Trapping/Sign: p = 1.000). Once again, no differences were apparent between methods when examining mammal ASR (HSD; Sign/Camera: p = 0.073; Trapping/Camera: p = 0.073; Trapping/Sign: p = 1.000) or reptile ASR (HSD; Sign/Camera: p = 0.635; Trapping/Camera: p = 0.989; Trapping/Sign: p = 0.743).
Figure 7.4 Comparison of (a) Survey species richness (SSR) and (b) Accumulative species richness (ASR) for (i) Total/All species, (ii) Target species (iii) Target mammal species (iv) Target reptile species. Generalised linear mixed models (GLMMs) were generated for the species richness measures with survey method as an explanatory variable. Site was included as a random variable. Differences between survey methods were tested for significance with post-hoc Tukey Honestly Significant Difference (HSD) test. Different letters indicate statistically significant differences.
7.3.3. **Probability of detection between methods**

Six species recorded sufficient data for calculation of POD; *Dasycercus blythi*, *Pseudomys desertor*, *Notomys alexis*, *Ctenotus pantherinus*, *Tiliqua multifasciata* and *Liopholis kintorei*. For all species model fit increased with the inclusion of method as an explanatory variable (Table 7.2). For all species except *P. desertor* and *C. pantherinus*, live trapping showed the lowest detection probability of the survey methods tested here whilst sign surveys were clearly the most effective for detecting *D. blythi*, *T. multifasciata* and *L. kintorei*. *D. blythi* was also detected relatively well by IR cameras. *N. alexis* had similar POD for both sign surveys and IR Cameras. *P. desertor* and *C. pantherinus* were detected most readily by IR cameras and least readily by sign surveys. Although the POD of *P. desertor* by live trapping was better than sign surveys it was also highly variable.
Table 7.2 Model outcomes and detection probabilities (±SE) for selected species across sampling methods.

<table>
<thead>
<tr>
<th>Model Selection</th>
<th>Probability of Detection</th>
<th>Live Trapping</th>
<th>Sign Surveys</th>
<th>IR Camera</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Model</strong></td>
<td><strong>AICc</strong></td>
<td><strong>Δ</strong></td>
<td><strong>Weight</strong></td>
<td><strong>R² (adj)</strong></td>
</tr>
<tr>
<td><strong>Mammal</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>D. blythi</td>
<td>80.53</td>
<td>0.00</td>
<td>0.99</td>
<td>0.43</td>
</tr>
<tr>
<td>Method</td>
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<td>8.93</td>
<td>0.01</td>
</tr>
<tr>
<td>P. desertor</td>
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<td>0.00</td>
<td>0.99</td>
<td>0.45</td>
</tr>
<tr>
<td>Method</td>
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<td>8.74</td>
<td>0.01</td>
</tr>
<tr>
<td>N. alexis</td>
<td>64.46</td>
<td>0.00</td>
<td>1.00</td>
<td>0.63</td>
</tr>
<tr>
<td>Method</td>
<td>NULL</td>
<td>82.78</td>
<td>18.32</td>
<td>0.00</td>
</tr>
<tr>
<td><strong>Reptile</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C. pantherinus</td>
<td>73.54</td>
<td>0.00</td>
<td>0.99</td>
<td>0.40</td>
</tr>
<tr>
<td>Method</td>
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<td>83.12</td>
<td>9.57</td>
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<tr>
<td>T. multifasciata</td>
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<td>0.75</td>
</tr>
<tr>
<td>Method</td>
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<td>19.42</td>
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</tr>
<tr>
<td>L. kintorei</td>
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<td>0.00</td>
<td>0.97</td>
<td>0.41</td>
</tr>
<tr>
<td>Method</td>
<td>NULL</td>
<td>59.43</td>
<td>6.77</td>
<td>0.03</td>
</tr>
</tbody>
</table>

7.3.4. Cost analysis

The capital costs associated with each survey method varied considerably. The cost of camera equipment was substantially higher ($18 000) than both live trapping ($5 000) and sign surveys ($100). However, the ongoing survey costs of camera surveys were lower ($2980/survey + $3040 for additional surveys and days) than both live trapping ($6300/survey + $1320 for additional survey days) and sign surveys ($5611/survey; See Appendix 12 for a detailed breakdown of costs). The calculated overall costs of each survey method over time showed live
trapping was initially and increasingly more expensive than sign surveys (Figure 7.5). Although IR camera surveys had the highest initial costs, live trapping became more expensive after four surveys and sign surveys after eight surveys as a result of larger ongoing survey costs. After 12 survey periods the total cost of each survey method varied considerably, with live trapping totalling $81 920, sign surveys $67 432 and IR cameras $56 800.

![Figure 7.5 Accumulated total survey costs of three comparative survey methods over 12 survey periods](image)

### 7.4. Discussion

My study is the first to directly compare the viability of three different survey methods to detect and monitor fauna across multiple taxa in a spinifex sand plains habitat. I found relatively strong support for the majority of my initial predictions, particularly those regarding the variation in capture rates in methods, the probability of detection of individual species and the detection of species richness across all surveys. Our initial prediction regarding detection of species richness in a single survey was partially supported with IR cameras detecting the highest richness for target mammal species. I also found partial
support for our cost analysis predictions due to the high initial capital expenditure but low ongoing costs associated with IR cameras. Although the costs over time followed our predictions IR cameras became the most economical quicker than initially expected.

7.4.1. Effect of varying deployment methods
My study showed captures rates were higher for mammals when using box traps whilst pitfalls were higher for reptiles. This result is expected as the standard bait utilised in box traps was designed to attract small mammals rather than reptiles. My results do highlight the importance of selecting live trapping methods/baits based upon the outcomes desired by the study (targeting species/groups vs general surveying methods). Variations in the success of live traps has also shown to vary between sites and across time, particularly in relation to environmental changes (e.g. Dickman et al. 2011; Pavey et al. 2008; Read et al. 2015b). I suggest implementing site and time specific pilot studies or the adaptive development of techniques based upon success to increase data collection in future studies.

Unlike live trapping, sign surveys provided a clear (and expected) consistent effect by deployment (experience level) for both mammals and reptiles with significantly more sign being recorded when a more experienced tracker was used (intermediate = 1263; novice = 419). Although not quantified in this study, the intermediate tracker consistently noted that they covered a greater proportion of the 2 ha grid than novice trackers, as they were able to identify sign more quickly. Previous studies examining the efficacy of sign surveys (e.g. Camargo-Sanabria & Mandujano 2011; Silveira et al. 2003) have shown considerable variations in detection depending on the area covered by trackers, which may have exacerbated the differences found here. Future research into
the effect of increased time allocation based upon experience level would clarify these findings and improve the application of sign surveys as a standardized method of species detection and monitoring. As expected and similar to the information on other taxa (e.g. Boitani & Powell 2012), the precision of data was also greater with increased experience levels. For example, novice trackers identified a greater number ‘small mammal’ tracks (intermediate = 4; novice = 82) whilst intermediate trackers identified more rodent (intermediate = 96; novice = 0) and dasyurid (intermediate = 43; novice = 6) tracks. The identifying features of rodent and dasyurid tracks are very difficult to discern (See Moseby et al. 2009c for details) and likely beyond the experience of novice trackers, which resulted in only small mammals being recorded by novice trackers and a reduction in the precision of data. Although these differences are clear and expected, many studies utilising sign surveys fail to report on the level of tracker expertise (e.g. Silveira et al. 2003) making assessment of accuracy difficult. Although not relevant to the analyses undertaken in this study, the potential additional social benefits of sign survey through the employment and/or involvement of local/indigenous trackers should also be considered during survey design (Crossing et al. 2016).

With the exception of small mammals at Newhaven, our results coincided with several previous studies (e.g. De Bondi et al. 2010; Paull et al. 2011) which documented greater success in capturing both mammals and reptiles using IR cameras with lures. The exception found in mammals at Newhaven may be due the greater detection of mammals overall at Newhaven (60/site/survey) compared to the borefields (14/site/survey). Although not tested directly, when abundances are higher the probability of capturing individuals is likely greater, regardless of lure presence, and would potentially override any positive
influence of the lure. This highlights an important variation in the effectiveness of lures in increasing small mammal detection across areas with varying abundance levels. More detailed examination across a broader range of sites with varying abundance levels would clarify this. In all other cases, our results coincided with our predictions that capture rates would increase with the use of lures. During many photo events animals became interested in the lure, which reduced movement and increased the number of photos taken, which improved the ability to identify certain species, particularly small mammals. Species specific avoidance or negative influence of lures has been noted in several other studies (Larrucea et al. 2007a; Larrucea et al. 2007b) but was not found to be an issue here, as no mammal species was detected considerably more (> 20 photo event difference) by cameras without lures than with lures. Notably, all IR cameras utilised in this study emitted a faint red light (semi-covert models) and may have deterred shy species therefore reducing their detection by all cameras overall and over-riding any avoidance behaviours due to lure presence (Meek et al. 2014). Further studies using fully covert cameras would clarify whether the models used here increased avoidance behaviours in shy species across all cameras deployed. These studies would also clarify whether the increased interest and subsequent precision in identification shown in this study was not at the cost of detecting more cautious species.

The only reptile species that showed a dramatic increase in the number of photo events on lured cameras was *C. pantherinus* (lure = 58; no lure = 5). A study investigating the foraging modes of *C. pantherinus* by Gordon et al. (2010) showed a reliance on olfactory cues to seek out food resources, which may have increased the attraction to the scent bait. Greater insight into the foraging modes of other small skinks, particularly *Ctenotus* species, may provide insight
into the findings of this study and aid in increasing capture rates in future studies. The use of lures on IR cameras in previous literature has predominantly targeted specific species/groups of interest (e.g. Fancourt 2014; Paull et al. 2011; Tanner & Zimmerman 2012). This study highlights the potential successful use of cameras and lures in non-specific faunal surveys.

7.4.2. Survey methods
Our research concurred with previous research that showed sign surveys were more proficient at detecting a broader range of species, particularly larger (non-target) species, such as cats, dingoes and kangaroos (Catling et al. 1997). However, the types of live traps used and low lying camera positioning of camera traps in this study would have created bias in their ability to survey larger species. By selecting different deployment methods (i.e. camera orientation or trap type) these methods may have broadened the detection of non-target species, as seen in other studies (e.g. Glen et al. 2013; Hamel et al. 2013), to similar levels of sign surveys.

In a single survey IR cameras were found to be more capable of detecting target small mammals than both live trapping and sign surveys. However, differences between methods were not apparent once richness was accumulated across all surveys. Notably, live trapping was the only method to detect *M. mus, S. youngsoni* and *P. hermannsburgensis* as both camera and sign surveys were unable to identify them to species level due to a lack of distinctive physical and sign characteristics. It is likely that although cameras and sign surveys identified a broad range of species quickly they were unable to continue detecting new species as they had reached the limit of their identification ability. Live trapping on the other hand would be likely to continue to identify
additional species, albeit at a slower pace, eventually equalling or potentially surpassing both cameras and sign surveys given time.

Although not significantly different, similar trends were also noted in reptiles with a more diverse group of species being identified irregularly by live trapping and cameras compared to a smaller group that was consistently recorded by sign surveys. The ability of sign surveys to detect a species is limited by the uniqueness of sign left. This is shown clearly as the majority of previous studies that have used sign surveys have focused on a single species or guild (e.g. Barea-Azcon et al. 2007; Bennison et al. 2014; Gompper et al. 2006; Southgate et al. 2005).

Examination of the probability of detection (POD) further supported the above point, as sign surveys performed well (> 70%) for all species except P. desertor and C. pantherinus, which do not leave distinctive sign. The PODs also showed inconsistent detection of species through live trapping and relatively low scores for all species tested (max = 0.50) while cameras performed well (> 0.65 POD) for all species except T. multifasciata (POD = 0.06) and L. kintorei (POD = 0.16). The low detection of T. multifasciata and L. kintorei may have been due to the reduced likelihood of stationary cameras detecting less mobile species (Burton et al. 2015). T. multifasciata are known to move between multiple small core areas of high use (Price-Rees et al. 2013) whilst L. kintorei show high site fidelity to their natal burrow systems (McAlpin et al. 2011). This lack of dispersal and movement would severely reduce IR cameras ability to detect the species unless they are placed strategically in areas identified as high use. The ability of cameras to detect less mobile species in general surveys is a clear drawback of their use (Burton et al. 2015) and should be considered in relation to sampling design and the species assemblages being sampled. Notably, this has also been
7.4.3. **Cost analysis**

Financial comparisons of survey methods indicated sign surveys were initially the most cost effective survey method; however, high ongoing survey costs (associated with personnel costs) quickly increased costs above that of IR cameras. Since IR cameras required personnel with minimal experience in the field, the ongoing field costs of surveys were low and therefore became the most cost effective after eight surveys. In the arid zone, longer term surveys are generally preferred as intermittent climatic changes create dramatic population fluctuations which drive species persistence over longer time frames (Dickman et al. 1999; Greenville et al. 2012; Lindenmayer et al. 2012b). Therefore, it is likely that cameras would be the most cost effective solution for many arid zone studies. The remoteness of arid areas makes the deployment of cameras for long periods without the need for constant personnel presence also very appealing. Given the extreme conditions, it is paramount to conduct pilot studies, as done here, to ensure cameras are deployed in a manner that will minimise failure in the field. Furthermore, while there are high capital costs associated with both camera and traditional methods, equipment is reusable between surveys and therefore costs may be spread over numerous surveys substantially reducing the longer-term costs.

7.4.4. **Management recommendations**

The design of fauna monitoring methods will always involve a trade-off between the scientific issue being addressed by the study and logistical and financial constraints. Based on my findings I recommend the increased use of infrared cameras for sampling of species richness across small mammals and reptiles in
spinifex sand plain habitat. This method provided the most consistent data across deployment techniques, sampled a high diversity of small mammal and reptile species, had relatively high detection probabilities for individual species and was found to be the most cost effective over time. Further experimentation on the use of different camera models and methods of camera deployment would be beneficial. The short pilot study conducted prior to this project tested only a single camera model and found a low lying vertical camera position provided the greatest number of detections/identification (J Molyneux 2012, unpubl.). Further exploration into species detection over longer periods and across different orientations and cameras models may help to further refine the use of cameras and broaden data collection even further.

Although IR cameras provided the best all round data across species, studies targeting specific species may benefit from the use of alternate methods such as sign surveys. For species which leave distinctive signs, such as *D. blythi* or *L. kintorei*, sign surveys may be a viable option for short term rapid monitoring as these provide quality data for the lowest financial expenditure, provided experienced personnel are used. The use of IR cameras can also be considered for single species studies. Previously they have proven very capable of detecting a variety of cryptic species (De Bondi et al. 2010; McDonald et al. 2015a; Welbourne et al. 2015) and used in conjunction with sign surveys can be placed strategically to areas where species are known to inhabit (i.e. runways, burrows, nest).

Although live trapping performed relatively poorly and was considered the most financially expensive when compared to the other methods in most analyses, it must still be considered as an important option in adequately sampling in the region. Morphological and reproductive data at both the individual and
population level such as body mass, breeding period and sex ratios are only obtainable using this method for many species. Furthermore, many of the smaller rodents and dasyurids could only be positively identified through detailed inspection of footpads or teeth making live trapping the only option.

Many monitoring projects currently use predominantly live trapping to survey sites and estimate small mammal and reptile presence and persistence (e.g. AWC 2011; Davies & Drew 2014; Dickman et al. 2001; Letnic et al. 2004). I believe these projects would benefit greatly from diverting some of the high costs associated with live trapping into purchasing IR cameras. IR cameras could be deployed alongside regular trapping regimes to complement and enhance data collection immensely whilst not substantially increasing ongoing field costs. The additional data would also help to provide information on the detectability of individual species by live trapping and how this changes over time.
CHAPTER 8

GENERAL DISCUSSION

Photo: Sunset at Newhaven Wildlife Sanctuary (A Molyneux)
8.1. **Overview**

Australia’s arid zone experiences considerable variability in ecosystem functioning over time that is thought to generally flow from bottom-up control in bust periods to top-down control in boom periods (Letnic & Dickman 2006; Meserve et al. 2003). Although the effects of rainfall variability are well documented (e.g. Greenville et al. 2012; Kwok et al. 2016; Read et al. 2012), not all species of fauna and flora show dramatic responses to rainfall. Specifically, there is a component of the biodiversity of arid Australia that shows seasonal responses in relation to habitat and other resources irrespective of rainfall (e.g. Pavey & Nano 2009). My study has contributed to developing our understanding of processes that influence fauna assemblages during the early bust phase. My key findings are:

- Large mammalian predators are influential to the occurrence of most nocturnal small mammal and reptile species during the onset of La Niña events,
- Climatic variation drives invertebrate occurrence with responses varying dramatically across orders,
- *Dasycercus blythi* utilise refuge patches with site-specific vegetation characteristics during periods of low rainfall,
- *D. blythi* actively select the open areas between vegetation for foraging despite the potential for increased levels of exposure to predators, and
- IR cameras provided the most consistent data on occupancy of sites by vertebrates, sampling across the broadest range of species and being the most cost effective method for long-term studies.

I detail each of these key findings in the sections below.
8.2. Persistence in a variable climate

Previous studies have shown that during periods of bust, the persistence of species is primarily limited by rainfall and primary production (Letnic & Dickman 2006; Meserve et al. 2003). My results showed little support for this. The presence of large mammalian predators was the most consistent factor influencing vertebrate’s abundances and diversity. Typically, the abundance of predators such as cats and foxes, is thought to be driven by fluctuations in rodent populations (Mahon 1999) which in turn are driven by rainfall (Greenville et al. 2012; Letnic et al. 2005). Although top up rainfall events during the study likely promoted the reproductive success of predators and thus resulted in greater predator presence, I failed to detect a similar response by most rodents. It is likely that several top up rainfall events experienced at Newhaven during the post boom decline helped support large predator populations longer, which in turn potentially suppressed the expected response by rodents (Paltridge 2005). Given the area consistently receives greater rainfall than other arid regions, even in periods of bust, it may also experience higher levels of predator numbers for longer periods following boom periods. This greater level of predation pressure is of particular concern for *Pseudomys hermannsburgensis*, which showed a dramatic decrease in occurrence once cats were detected at over 50% of sites. Furthermore, given that climate change predictions indicate boom and bust cycles will become more intense in the future (Intergovernmental Panel on Climate Change [IPCC] 2014) the long term effect of this suppression is potentially profound (Greenville et al. 2016).

In contrast to vertebrates, invertebrate abundances and diversity fluctuated consistently with climate variables. These results coincide with several previous findings in the region (e.g. Langlands et al. 2006; Popic et al. 2013); however,
subtle ordinal specific responses were also detected. The differences in responses found here are potentially under-estimated due to the coarse level of classification (ordinal) and the potential influence of additional factors such as the degree of flowering/seeding and varying levels of soil moisture that were not examined. As invertebrates play key roles in many aspects of ecosystem functioning (Prather et al. 2013), gaining a greater understanding of the dynamics in this diverse group would not only fill a large gap in our current understanding in arid zone ecosystems, but is likely to contribute substantially to our ability to support and manage the system longer term.

Fire and the amount of vegetation cover were not found to be important drivers of species persistence in this study, except in the cover-reliant species *Pseudomys desertor*. Considering the consistent historical occurrence of both human induced fire (Latz & Green 1995) and wildfires (Burrows et al. 2006) in the landscape, many species would likely have developed a natural resilience to its occurrence which has aided in buffering against its effects (Nano et al. 2012). Although my results indicated fire management plays a very limited role in influencing the overall diversity and occurrence of many species, its implementation as a land management tool would have a profound effect on certain species that show greater sensitivity (such as *P. desertor*). Studies such as this one, that clearly identify fire sensitive species can help land managers target fire management to provide valuable protection from larger fires, through fire breaks, whilst avoiding areas and minimising the effect on sensitive species. Like several previous studies, this study supports the idea that fire itself does not create broader biodiversity but instead can be used to protect areas that do (e.g. Pastro et al. 2011).
Notably, despite the large proportion of pastoral land in Australia’s arid zone both study sites examined here have been subjected to very little agricultural influence. When examining species across their broader distribution understanding the potential additional effects of agriculture is important to ensuring appropriate management within individual sites is undertaken (e.g. McGregor et al. 2014).

8.3. Identifying species specific requirements for Dasycercus blythi

As one of the largest remaining native mammalian predators in the region, it is essential that we understand the dynamics that influence Dasycercus blythi populations. My study showed clear contraction of individual D. blythi populations when the level of rainfall the previous year reduced. Northern populations (Newhaven) showed a less dramatic contraction which is likely due to the higher rainfall, particularly during winter, which is experienced in areas of Australia’s arid zone that experience monsoonal influences. With climate change predictions estimating longer, drier periods of bust (IPCC 2014) populations which have greater monsoonal influences may prove important to ensure the persistence of D. blythi over time (Cere et al. 2015). In contrast, such long term climatic changes are likely to increase the pressures in southern populations and further restrict these isolated populations, potentially making them increasingly vulnerable to stochastic events and ultimately local extinction (Bürger & Lynch 2013).

In this study, areas where D. blythi persist during periods of low rainfall showed subtle variations in vegetation characteristics between sites making broad scale identification of D. blythi refuge areas difficult. Although based on a single site (Newhaven), I did identify several important habitat components at a finer scale.
that may help to support populations, particularly in regards to reproductive success, including shrub cover, termites and great desert skink burrows. Determining whether these components consistently influence the success of *D. blythi* populations across its distribution more broadly will be important in determining the environmental components necessary for the persistence of the species.

At a microhabitat scale, this study found *D. blythi* actively select the open areas between vegetation when foraging. While this strategy potentially increases access to food resources, it also seemingly increases exposure to introduced predators, such as cats and foxes. Regardless, *D. blythi* occur rarely in the scats of introduced predators (Paltridge 2002; Spencer et al. 2014) indicating it is capable of avoiding predation whilst utilising these open areas. The highly responsive nature of *D. blythi* found in this (See Appendix 8) and other studies (Woolley 1990) may indicate it relies more on their ability to detect predators and retreat to shelter sites (i.e. burrows or *Triodia* clumps) rather than selecting areas that provide protection. However, studies have found evidence that lower levels of cover from fire combined with high predator abundance is likely to result in the exclusion or removal of individuals from these areas (Greenville et al. 2016; Körtner et al. 2007). It is likely the predatory avoidance strategies utilised by *D. blythi* are limited when conditions are unfavourable (as described previously) and should be an important consideration when managing landscapes, particularly in areas of refuge where populations are most vulnerable.
8.4. Improving monitoring methodology

In Australia’s arid zone, land managers face considerable challenges in collecting accurate and adequate biodiversity information. Spatial scales are vast resulting in large climatic variations, funds are normally limited, and it can be difficult to adequately sample across a diverse range of species (Byrne et al. 2008; Letnic et al. 2011; Morton et al. 1995; Perry & Goodall 1979; Smith & McAllister 2008). Not surprisingly, this study showed that during periods of low rainfall current live trapping practices did not detect many of the species present in the landscape or prove financially practical. However, live trapping did perform better in terms of the level of taxonomic classification and accuracy, and was the only method that was capable of identifying all potential species, particularly small mammals such as dunnarts. Sign surveys provided the most cost effective method over short time periods, but failed to detect many species as a consequence of some signs being indistinguishable at the species level. Overall, infrared cameras were most cost effective over longer time periods and enabled sampling across a broad range of species (both target and non-target). The implementation of multiple methods in research programs has previously been suggested to improve the implementation of adaptive management (Lindenmayer et al. 2012a), as it can result in the targeted use of methods to meet specific management goals across and within individual projects. Based upon the findings here it is clear that in the spinifex sand plains habitat, implementing a greater variety of survey methods (i.e. camera trapping and sign surveys) would improve the accuracy and amount of data collected during surveys. Furthermore, managers and researchers can utilise these methods to identify and target live trapping to areas where capture rates are likely to be higher (as done here for *D. blythi*). Ensuring biodiversity monitoring programs
utilise the most effective strategies is paramount and should be considered carefully during project design (Lindenmayer et al. 2012a). Continuing to conduct comparative research across the suite of methods available is paramount, not only for the data they provide but also to account for the limitations of each. With increasing development in technology and statistical analysis these limitations are constantly evolving, highlighting the importance of continual review and refinement.

8.5. A pathway to better management

Developing better strategies for effective management to improve biodiversity condition is a consistent theme in conservation ecology, including in the literature relevant to the Australia’s arid zone (e.g. Doherty et al. 2015b; Lindenmayer et al. 2012a; Morton et al. 2011). Many of these studies have identified the lack of broad-scale collaboration to be one of the greatest inhibitors to the successful implementation of biodiversity conservation (Lindenmayer et al. 2012a). In Australia’s arid zone this is particularly relevant considering the large extent of the area as well as a wide variety of land owners and land use objectives (Commonwealth of Australia 2009). As many fauna species occur in small isolated populations and are particularly hard to detect on a regular basis (Bos & Carthew 2001; Dickman et al. 2001; Morton 1978; Morton & James 1988), many landowners are often responsible for a single isolated population. This often results in limited application of monitoring and management across broader scales. Furthermore, restrictions in resources such as time, equipment and people, often leads management to implement only a single method over a relatively small spatial scale (Curtis & De Lacy 1998). Improving communication, data sharing and the development of management strategies that incorporate the limitations and requirements of various
landowners across land tenures would provide better outcomes for species more broadly. Strategies such as the creation of communal reporting systems (Eyre et al. 2011) and/or the introduction of financial incentives for reporting and monitoring (Windle & Rolfe 2008) may help to facilitate these improvements. In addition, the creation of a centralised body to collate this information and provide specific recommendations tailored to the different management groups would ensure a greater cohesiveness in regional outcomes.

This study has helped to identify how these managers can implement better monitoring across broader scales and identified how a combination of broad and fine scale studies can provide more rounded management strategies. This study also highlighted the clear lack of knowledge regarding invertebrates in Australia’s arid zone and the need to improve our identification and level of understanding in this group.

This study showed considerable differences in efficacy between methods and highlighted the importance of developing monitoring programs that reflect these differences. Notably, this study focused on a period of lower rainfall when species detection is more challenging (Letnic & Dickman 2006; Letnic et al. 2005; Lindenmayer et al. 2012b). Although live trapping failed to detect many species in this study, it has proven to be very successful in monitoring populations during periods of higher rainfall and in other locations (e.g. AWC Australian Wildlife Conservancy [AWC] 2016a; Dickman et al. 2010; Greenville et al. 2013; Kelly et al. 2013; 2005). Given the large scale fluctuations in the detectability of species, monitoring programs may benefit by altering the way they sample over time when assessing biodiversity more broadly (i.e. by using live trapping during high rainfall periods and IR cameras during low rainfall periods). This would reduce the ongoing financial costs of monitoring whilst still
maintaining data integrity. This would further enable the redirection of funds across a broader spatial scale or alternatively fund research that focuses on the specific goals aimed at identifying areas or components in the landscape that support individual populations or species (as done here for *D. blythi*). By identifying these components we can better target management strategies such as fire regimes, predator removal and species specific monitoring.

By focusing some of my monitoring on *D. blythi* I was able to identify several areas and habitat components that were important for *D. blythi* and increased our understanding on how it persists in landscapes where several other species of similar size have gone extinct (Johnson 2006; Woinarski et al. 2014). My findings found little consistency between sites in the vegetation characteristics in areas that support populations during dry periods making broad scale identification of these areas difficult. Due to the uniqueness of *D. blythi* tracks, I highly recommend the use of sign surveys and IR cameras to identify refuge areas that support populations in low rainfall periods in order to target management actions more effectively. My study showed that supporting the retention of shrub cover and healthy populations of termites and great desert skinks in these areas may further help to promote the persistence of *D. blythi* in low rainfall years. I further propose management should implement targeted large mammalian predator control in these areas and regular monitoring of predator diets to ascertain shifts in prey preference towards *D. blythi*. Although fire was not shown to influence habitat selection in this study, preserving areas where proximity to cover can be maintained may help to buffer against increased predation where it occurs.

Gaining greater knowledge of the behaviour of *D. blythi*, particularly in relation to reproduction and foraging, would further clarify the resource requirements of
the species. With substantial changes expected in arid Australia as a consequence of climate change, ensuring land owners that manage individual populations continue to collaborate will help to identify the differences in changes that are occurring across numerous isolated populations and will give a better measure of the future of *D. blythi* overall.

8.6. Conclusion

This thesis has investigated various approaches to monitoring the persistence of species in the spinifex sand plains of arid Australia in a period following the post boom decline. In general, I found large mammalian predators to be the primary driver of vertebrate occurrence whilst climate proved more important for invertebrates. I found *D. blythi* respond to variations in long-term rainfall. Populations became less widespread when rainfall the previous year fell below 400 mm. During low rainfall periods *D. blythi* actively utilised open areas, which likely improved foraging success; however, would also increase exposure to predators. Persistence by *D. blythi* during these periods was facilitated by resource rich patches that were potentially used to promote greater reproductive success. During these periods, when population numbers are low, accurate detection across a diverse range of species was difficult. I found IR cameras to be the most effective and efficient in monitoring population’s longer term, although many species could only be identified through live trapping as it enabled more detailed examination. This thesis has added to our knowledge on species assemblages and effectiveness of management in Australia’s spinifex sand plains habitat and provided recommendations to refine its management strategies in the future.
Photo: Heading home with field work complete! (A Molyneux)
**Appendix 1** List of all candidate models for mammal and reptile diversity and individual abundance. ^ indicates offset variable; * indicates random variable ◊ indicates the abundance measure included (reptile abundance in mammal models and mammal abundance in reptile models).

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Appendix 2 All mammal candidate model outcomes. All models with considerable support ($\Delta$AICc < 4) are highlighted in bold.

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Appendix 3

All reptile candidate model outcomes. All models with considerable support ($\Delta$AICc < 4) are highlighted in bold.

### Reptile Richness

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**Appendix 5** All invertebrate candidate model outcomes. All models with considerable support (ΔAIC < 4) are highlighted in bold.

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Appendix 6 Site layout of live traps during annual surveys at Newhaven Wildlife Sanctuary (Moore 2013).
Appendix 7 Location of live trapping (closed) and sign survey (open) sites at Uluru Kata–Juta National Park between 2000 and 2011.
Appendix 8 Additional methods tested to assess microhabitat selection by *D. blythi* in the spinifex sand plains at Newhaven Wildlife Sanctuary.

In 2013, several different methods were attempted to assess habitat selection and predator awareness in *D. blythi*. Across four survey periods I tested the use of GPS transmitters, VHF tracking, giving-up-density experiments and chemiluminescent tagging with the following results.

GPS and VHF tracking

Seven *D. blythi* were captured and released with GPS collars (Model: EP – 3.3; Ecotone, Poland) in 2013. The data from GPS collars were designed to remotely download to fixed base stations (Model: EP - BS11; Ecotone, Poland) located in the area surrounding their capture. Although I had some successful downloads (43 locations for one male and 12 locations for one female) all other collars failed to download to base stations in the area. In May 2013, field tests of equipment showed inconsistent range detection and/or transmission between base stations and collars. After consulting with the manufacturers I conducted further field tests by deploying collars with both GPS and VHF capabilities (Model: V1G102A; Sirtrack, Havelock North, New Zealand). These tests confirmed collar data was not downloading to base stations. Due to the time and financial constraints and the availability of other GPS options at the time they were deemed unsuitable for this project.

During our assessment of GPS collars I also investigated the potential standalone use of VHF tracking to assess habitat selection by *D. blythi*. Tracking was completed successfully throughout the day (when individuals were inactive and in burrows); however, as personnel approached individual *D. blythi* at night (when animals were active) they were highly responsive and
immediately retreated to burrows when approached within 15 meters. As
greater accuracy was needed VHF was also deemed unsuitable.

Giving-Up-Density (GUD) experiments
I conducted giving-up-density (GUD) experiments, which has been successful in
assessing predator awareness by small mammals previously (e.g. Bleicher &
Dickman 2016; Stokes et al. 2004). Although initially *D. blythi* were readily
detected at GUD stations, detection rates fell dramatically over subsequent
surveys resulting in insufficient data for analysis. The lack of detection was
likely due to a reduction in abundance and density across the site more
generally as it corresponded with a similar reduction in live trapping and IR
camera detection rates.

Chemi-luminescent tagging
Five *D. blythi* were released with small chemi-luminescent tags following the
protocols for data collection and attachment described by Bos and Carthew
(2007b) and Philp (2011). As seen with the VHF tracking, *D. blythi* were highly
responsive to the presence of trackers and the tag. Two individuals shed the tag
within five minutes of release; one individual shed the tag in a burrow whilst
the other was seen actively grooming the tag off under spinifex. The remaining
three *D. blythi* released with tags all displayed negative responses when
trackers came within 15 meters, uniformly resulting in the individual retreating
to the nearest burrow or large clump of spinifex and remaining there until
trackers withdrew. As sufficient detail on *D. blythi* behaviour was unable to be
collected at distances over 15 meters and trackers could not be confident that
individuals were displaying completely natural behaviours when up to 20
meters away this method was deemed unsuitable.
Appendix 9 Maps showing the four pathways of spool lines identified as areas of intense use. Line indicates spool pathway; X indicates start of spool line; orange circle indicates burrow and grey circle indicates area of intense use.
Appendix 10 Total captures of mammal species/groups by survey method and deployment technique. Percentage of sites with records indicated in parenthesis.

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<td></td>
</tr>
<tr>
<td></td>
<td>2(25)</td>
<td>1(13)</td>
<td></td>
</tr>
<tr>
<td><em>Pseudomys desertor</em></td>
<td>5(38)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Canis dingo</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>41(100)</td>
<td>10(38)</td>
<td></td>
</tr>
<tr>
<td><em>Tachyglossus aculeatus</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>3(13)</td>
<td>1(13)</td>
<td></td>
</tr>
<tr>
<td><em>Mus musculus</em></td>
<td>8(50)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Sminthopsis youngsoni</em></td>
<td>1(13)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Dasycercus blythi</em></td>
<td>1(13)</td>
<td>2(25)</td>
<td>81(100)</td>
</tr>
<tr>
<td><em>Oryctolagus cuniculus</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>6(38)</td>
<td>5(63)</td>
<td></td>
</tr>
<tr>
<td><em>Macropus rufus</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>27(75)</td>
<td>5(38)</td>
<td></td>
</tr>
<tr>
<td><em>Pseudomys hermannsburgensis</em></td>
<td>23(50)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Notomys alexis</em></td>
<td>2(25)</td>
<td></td>
<td>95(100)</td>
</tr>
<tr>
<td>Dunnart sp.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>P. hermannsburgensis/M. musculus</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dasyurid</td>
<td>43(100)</td>
<td>6(38)</td>
<td></td>
</tr>
<tr>
<td>Rodent sp.</td>
<td>7(38)</td>
<td></td>
<td>96(100)</td>
</tr>
<tr>
<td>Small mammal</td>
<td>4(38)</td>
<td>82(100)</td>
<td>64(100)</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td>47</td>
<td>2</td>
<td>557</td>
</tr>
</tbody>
</table>
**Appendix 11** Total captures of reptile species/group by survey method and deployment technique. Percentage of sites with records indicated in parenthesis.

<table>
<thead>
<tr>
<th>Species/Group</th>
<th>Box</th>
<th>Pitfall</th>
<th>Intermediate</th>
<th>Novice</th>
<th>Lure</th>
<th>No Lure</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Live Trapping</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Sign Surveys</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Camera Traps</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pogona vitticeps</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Tiliqua multifasciata</em></td>
<td></td>
<td></td>
<td>34(100)</td>
<td>3(38)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Ctenotus grandis</em></td>
<td>2(25)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>C. hanloni</em></td>
<td>4(38)</td>
<td></td>
<td>18(88)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>C. pantherinus</em></td>
<td></td>
<td></td>
<td>9(50)</td>
<td></td>
<td>58(88)</td>
<td>5(38)</td>
</tr>
<tr>
<td><em>C. schomburgkii</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Eremiascincus fasciolatus</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Liopholis kintorei</em></td>
<td></td>
<td></td>
<td>21(63)</td>
<td>34(88)</td>
<td>2(25)</td>
<td>2(13)</td>
</tr>
<tr>
<td><em>Strophurus elderi</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>S. jeanae</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Lerista bipes</em></td>
<td></td>
<td></td>
<td>2(25)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Ctenophorus isolepis</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>14(63)</td>
<td>12(25)</td>
</tr>
<tr>
<td><em>C. nuchalis</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1(13)</td>
</tr>
<tr>
<td><em>Moloch horridus</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>16(88)</td>
<td></td>
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<tr>
<td><em>Notoscinctus ornatus</em></td>
<td>8(75)</td>
<td></td>
<td></td>
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<tr>
<td><em>Varanus giganteus</em></td>
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<td></td>
<td>90(100)</td>
<td>2(13)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>V. eremius</em></td>
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<td></td>
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<td></td>
<td>4(38)</td>
<td>5(50)</td>
</tr>
<tr>
<td><em>V. gouldii</em></td>
<td></td>
<td></td>
<td>8(25)</td>
<td></td>
<td>2(25)</td>
<td></td>
</tr>
<tr>
<td><em>V. brevicauda</em></td>
<td></td>
<td></td>
<td>3(25)</td>
<td></td>
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<td></td>
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<tr>
<td><em>Lerista spp.</em></td>
<td>15(75)</td>
<td></td>
<td>7(50)</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Species/Group (cont.)</td>
<td>Live Trapping</td>
<td>Sign Surveys</td>
<td>Camera Traps</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>-----------------------</td>
<td>---------------</td>
<td>--------------</td>
<td>--------------</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Box</td>
<td>Pitfall</td>
<td>Intermediate</td>
<td>Novice</td>
<td>Lure</td>
<td>No Lure</td>
</tr>
<tr>
<td>Legless Lizard</td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
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<tr>
<td>Dragon</td>
<td>1(13)</td>
<td></td>
<td>1(13)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Goanna</td>
<td>280(100)</td>
<td>37(88)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Large snake</td>
<td>5(38)</td>
<td></td>
<td>3(13)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Skink</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>4(25)</td>
</tr>
<tr>
<td>Small snake</td>
<td>28(75)</td>
<td>16(75)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lizard</td>
<td>215(100)</td>
<td>77(100)</td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Reptile</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1(13)</td>
</tr>
<tr>
<td>TOTAL</td>
<td>6</td>
<td>48</td>
<td>706</td>
<td>180</td>
<td>96</td>
<td>28</td>
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</table>
Appendix 12 Detailed costings for all survey methods assessed. Multiple surveys were calculated by multiplying ST1 by the number of survey of interest.

<table>
<thead>
<tr>
<th>Survey requirements</th>
<th>Live Trapping</th>
<th>IR Cameras</th>
<th>Sign Surveys</th>
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<tbody>
<tr>
<td># survey days (L)</td>
<td>9</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td># personnel (P)</td>
<td>2</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td># experience pers. (R)</td>
<td>1</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>Data entry hours (H)</td>
<td>3</td>
<td>12</td>
<td>3</td>
</tr>
<tr>
<td>Additional set up days (S)</td>
<td>2</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td># extra surveys (Z)</td>
<td>0</td>
<td>1</td>
<td>0</td>
</tr>
</tbody>
</table>

**Capital costs**

| Equipment (E) | 5 000 | 18 000 | 100 |

**CAPITAL TOTAL**

<table>
<thead>
<tr>
<th>Ongoing survey costs</th>
<th>Live Trapping</th>
<th>IR Cameras</th>
<th>Sign Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>Car Hire</td>
<td>1 320</td>
<td>540</td>
<td>800</td>
</tr>
<tr>
<td>130L + 150</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Accommodation</td>
<td>240</td>
<td>60</td>
<td>120</td>
</tr>
<tr>
<td>15P x (L-1)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food</td>
<td>900</td>
<td>300</td>
<td>500</td>
</tr>
<tr>
<td>50P x L</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wages</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Survey L(50 x 8 hrs x R)</td>
<td>3 600</td>
<td>1 200</td>
<td>4 000</td>
</tr>
<tr>
<td>Data 30H</td>
<td>90</td>
<td>600</td>
<td>90</td>
</tr>
<tr>
<td>Consumables</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Rate of replacement</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>0.01E</td>
<td>50</td>
<td>180</td>
<td>1</td>
</tr>
</tbody>
</table>

**Survey sub-total 1 (ST1)**

<table>
<thead>
<tr>
<th>Live Trapping</th>
<th>IR Cameras</th>
<th>Sign Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>6 300</td>
<td>2 980</td>
<td>5 611</td>
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</tbody>
</table>

**Set up days**

<table>
<thead>
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<th>Sign Surveys</th>
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<tbody>
<tr>
<td>1 320</td>
<td>660</td>
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</tbody>
</table>

**Extra surveys**

<table>
<thead>
<tr>
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<th>IR Cameras</th>
<th>Sign Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>2 380</td>
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</tbody>
</table>

**Survey sub-total 2**

<table>
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<th>IR Cameras</th>
<th>Sign Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 320</td>
<td>3 040</td>
<td>0</td>
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</tbody>
</table>

**SURVEY TOTAL**

<table>
<thead>
<tr>
<th>Live Trapping</th>
<th>IR Cameras</th>
<th>Sign Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>7 620</td>
<td>6 020</td>
<td>5 611</td>
</tr>
</tbody>
</table>

**TOTAL (single survey)**

<table>
<thead>
<tr>
<th>Live Trapping</th>
<th>IR Cameras</th>
<th>Sign Surveys</th>
</tr>
</thead>
<tbody>
<tr>
<td>12 620</td>
<td>24 020</td>
<td>5 711</td>
</tr>
</tbody>
</table>


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