Reintroduction Biology of the Eastern Barred Bandicoot

Amy L. Winnard

Submitted in total fulfilment of the requirements of the degree of Doctor of Philosophy

August 2010

Department of Zoology
The University of Melbourne
Abstract

Worldwide, species are being driven to extinction due to anthropogenic activities that have led to habitat loss and modification, introduced species, over exploitation, pollution and climate change. As a result, species are now being reintroduced back into their former ranges, but many of these reintroductions fail. The eastern barred bandicoot *Perameles gunnii*, a threatened marsupial restricted to lowland grasslands in south-eastern Australia, has had many failed reintroduction attempts. This study aimed to improve the success of future eastern barred bandicoot reintroductions and expand our knowledge of this species’ ecological requirements to ensure its long term survival. I reviewed all previous eastern barred bandicoot reintroductions, determining how best to monitor bandicoots, analysing data from two different populations to find out the regulatory processes acting upon them and, examined how habitat quality influences the behaviour of bandicoots.

Since 1989 the eastern barred bandicoot has been reintroduced into eight sites: Woodlands Historic Park, Hamilton Community Parklands, Mooramong, Floating Islands Nature Reserve, Lake Goldsmith Wildlife Reserve, Lanark, Cobra Killuc Wildlife Reserve, and Mt Rothwell. All reintroduced populations initially increased, but declines were observed during the mid to late 1990s, culminating in extinctions at many sites. Although the exact cause of these declines is not known, a combination of drought and predation by red foxes *Vulpes vulpes* and cats *Felis catus* is thought to be responsible.

Monitoring threatened species is essential for determining growth trajectories and the response of populations to any management actions undertaken. I compared the effectiveness of three low-cost monitoring techniques in a low density, declining population, and in a relatively high density population. Trapping was suitable for monitoring only the high-density population, whilst spotlighting detected few bandicoots in both populations. Foraging dig counts were readily identified and counted at both reserves, with the density of digs being greater in the winter months. Using these three techniques together increased the probability of bandicoot detection, and identified the growth trajectory of the population and spatial occupation of the reserve.

Using data collected from live trapping, I analysed a 17-year data set of a reintroduced population not protected by a predator-barrier fence. This population showed some evidence of an Allee effect. Two measures of fecundity showed negative density
dependence, whilst high air temperature negatively affected breeding and litter size. The main cause of population regulation at this site is likely to be predation by introduced red foxes, exacerbated by a prolonged drought in the mid 1990s. I also analysed a smaller data set (2.5 years) collected from a newly-reintroduced population. This population also showed some evidence of an Allee effect and fecundity was negatively affected by air temperature. If this is a true Allee effect, then this reserve may need to be managed as a metapopulation.

In a recently reintroduced population of bandicoots, I found that soil compaction was the only habitat variable that influenced where bandicoots foraged. When the population had become established, soil compaction along with grass cover and distance to cover also became important. I also discovered that wild-bred bandicoots occupied smaller areas than founder bandicoots, probably due to a trade-off between increasing male-female interactions and decreasing male-male aggressive interactions in this growing population. The larger areas occupied by founders is probably due to males attempting to increase mating opportunities with patchily distributed females soon after reintroduction, then retaining their larger ranges once the population became established by dominance over younger, wild-bred animals.

In a reserve protected from foxes by a predator-barrier fence, but that had become severely overgrazed by macropod species, I found that bandicoots used the habitat in the same way as found in other studies in fox free areas. However the scale of their habitat use was much larger, highlighting the behavioural flexibility of eastern barred bandicoots in response to a structurally simplified habitat, and further demonstrates the importance of fox-free areas in reintroduction site selection.

The main management recommendations born from this thesis include: only releasing eastern barred bandicoots into fox-free areas, down grading the importance of ‘perceived’ habitat quality, releasing captive-bred animals in the cooler months when high quality or more food is readily available and implementing monitoring regimes, specific to each site, that can be used to obtain unbiased estimates or indices of density at all reintroduction sites. If the Recovery Team abides by these recommendations, then the success of bandicoot reintroductions will improve, and this species, that has the necessary qualities for successful reintroduction, should one day be taken off the critically endangered list.
Declaration

This is to certify that:

(i) the thesis comprises only my original work towards the PhD except where indicated in the Preface

(ii) due acknowledgement has been made in the text to all other material used

(iii) the thesis is less than 100,000 words in length, exclusive of tables, maps, bibliographies and appendices

______________________________  ______________________________
Signature                                      Date
Preface

All data presented were collected solely by the author with the exception of: monitoring data presented in Chapter 4, which was collected by the Department of Sustainability and Environment and its precursors, and winter 2009 monitoring data in Chapters 5 and 6 which was collected by Richard Hill of the Department of Sustainability and Environment.

Chapter 2 was published in Pacific Conservation Biology in 2008 and co-authored with my supervisor, Graeme Coulson, however, I am the senior author and principal contributor in this paper.
Acknowledgements

It’s taken five long, but enjoyable years to complete this thesis and I am indebted to many people for their help over the years with ideas, field work and comments on various versions of chapters. Being part of a recovery team meant that I was never short of people to support me, but first, before I get into a long list of names of people to whom I will be forever grateful to, I must thank the man responsible for starting me off on this roller coaster ride, and who has been with me every step of the way. That is, of course, my supervisor Graeme Coulson. I will be forever grateful to Graeme for taking me on as a student when all he really knew about me was that I was an undergrad from The University of Aberdeen, with an interest in threatened species management. I thank Graeme for introducing me to the eastern barred bandicoot and supporting me throughout this PhD. If it were not for his support and endless discussions and ideas, then this thesis would never have come to fruition.

The second person I must thank is Richard Hill, the convener of the Eastern Barred Bandicoot Recovery Team. Richard not only assisted me on many trap sessions at Hamilton Community Parklands, but is also the reason that I managed to change my candidature from part time to full time, due to his amazing ability to source money. This ability led to funding for half of my scholarship (of which the second half was funded by the Department of Zoology and Alfred Nicholls award) as well as almost all the funding I required to carry out this project.

Funding and scholarship money was provided by the Alfred Nicholls Award, Department of Sustainability and Environment, Department of Zoology (The University of Melbourne), Glenelg-Hopkins CMA and Parks Victoria and field work was carried out under DSE research permits 10003881, 10004897 and animal ethics permits 05186 and ZV07010.

As this PhD involved working at five locations all with different owners and managers, I have an endless list of people to thank. Firstly I need to thank Jasmine Ferguson for teaching me how to trap bandicoots and conduct habitat analysis. At Woodlands Historic Park I thank Parks Victoria ranger in charge, Fiona Smith, for permitting me to use the facilities, rangers Katrina Lovett and Susanne Coates and all volunteers including Marissa Parrot, Michelle Wilson, Ray McMahon for helping with setting up trapping grids, early morning trap checking, spotlighting and staying positive despite never catching a bandicoot.
Thanks also to all the other rangers at Organ Pipes, Parks Victoria that offered their assistance on occasion and kept me entertained with their banter.

At Hamilton Community Parklands I thank Richard Hill, Yvonne Ingeme and Reeto Zolinger; the fastest walkers I have ever come across. Barry ‘Buzz’ Carr who was always more than willing to help out and permitted me to use his quad bike, so I could conduct field work quicker. Thanks also to volunteers Julian Cooke and his work for the dole crew, Lauren Eddy, Bruce Ferguson, Sarah Garnick, Gavin Lewis, Megan Lutton, Cecilia Myers, Rob Reed, Justine Smith and Mandy Watson

Thank you to Jim O’Brien, DSE for answering all my Mooramong questions and supplying me with all the data he had collected over many years of trapping Mooramong bandicoots. Thanks also Jim for finding time in your hectic schedule to come out to Mooramong with me to find star pickets from years gone by. Thanks to David Coutts for having the brilliant idea that I should analyse the Mooramong dataset. There were many occasions that I wished you hadn’t suggested that, but now the chapter is written, I’m glad you did. Thanks also to Ian Waller of Mooramong for your conversations and answering my questions.

Thank you to Nigel Sharp and Rosemary Etherton of Mt Rothwell for permitting me to live at your amazing, but freezing reserve for five weeks. Paul Mervin and Annette Rypalski for keeping me entertained during my stay and helping out with field work whenever you had a spare moment, and Jemma Cripps, Dave Holmes, Justin Smith, Claire Saxby, Sumi Vasudevan, and Madi Yewers for their assistance with my field work.

Thanks also to Peter Courtney from Melbourne Zoo, for your support and assistance, the vets for teaching me how to microchip eastern barred bandicoots and for making time to figure out how best to attach a radio-transmitter to an eastern barred bandicoots tail and to the keepers for keeping an eye on all the animals in my radio-transmitter trial. I should also thank Kirsten Long for her long distance teaching on how to attach a radio-transmitter to a bandicoot’s tail.

Thank you to all the other volunteers that offered their time to help with trapping at all reserves. I may have forgotten your names but I will never forget the help you provided. I hope the reward of seeing one of Victoria’s most endangered mammals was worth it.

My deepest gratitude goes to Roger and Jacqui Brown who welcomed me into their family and offered me a warm bed and many delicious home cooked meals whilst I conducted...
field work in Hamilton. Their generosity was endless, despite them not knowing me from a bar of soap when I first arrived on their door step. I must also thank David Middleton; an old university friend of the Brown’s, who is the reason I stayed with them, instead of the lonely, local caravan park.

Many thanks to all the people I inundated with numerous questions about each reintroduction site so that I could compile my review article (Chapter 2); Peter Goldstraw, Joel Little, Katrina Lovett, Steve McDougall and Jim O’Brien. I am also grateful to all the people who have read various drafts of chapters over the years; Ami Bennet, Tim Bernard, Nat Briscoe, Greta Frankham, Sarah Garnick, Richard Hill, Tim Jessop, Katrina Lovett, Michael Lynch, Jen Martin, Jess Roberts, Alan Robley, Julian Di Stefano and Charles Todd. The Conservation and Climate change lab group also deserves a huge thank you for all your ideas and suggestions over the years, as well as David Paul for introducing me to Adobe Illustrator and Ken Grimes for permitting me to use his bandicoot north arrow in my figures.

A huge thank you to Graham Hepworth and Julian Di Stefano for the many hours they spent with me, teaching me statistics. I learnt a lot from you both and numbers don’t seem that scary to me anymore.

Last but certainly not least I would like to thank my little family. My amazing partner, Jaco Coetsee, for all his support over the years, for helping with field work and for fixing my computer problems (which I’m sure felt like they were never ending), and to my unborn baby for providing me with the ultimate thesis submission deadline!
## Contents

Abstract .................................................................................................................. i  
Declaration ............................................................................................................. iii  
Preface .................................................................................................................... v  
Acknowledgements ................................................................................................ vii  
Contents .................................................................................................................. xi  
List of tables ........................................................................................................... xiii  
List of figures .......................................................................................................... xv  

**Chapter 1: General introduction ................................................................. 1**  
Reintroduction biology .......................................................................................... 3  
Eastern barred bandicoots ..................................................................................... 4  
Aims ......................................................................................................................... 5  
Thesis outline .......................................................................................................... 6  

**Chapter 2: Sixteen years of eastern barred bandicoot Perameles gunnii**  
reintroductions in Victoria: a review ............................................................. 9  
Abstract .................................................................................................................. 11  
Introduction ........................................................................................................... 12  
Reintroduction Sites ............................................................................................... 14  
  1) Woodlands Historic Park ................................................................................. 14  
  2) Hamilton Community Parklands .................................................................... 18  
  3) Mooramong .................................................................................................... 21  
  4) Floating Islands Nature Reserve .................................................................... 24  
  5) Lake Goldsmith Wildlife Reserve ................................................................. 26  
  6) Lanark ............................................................................................................ 29  
  7) Cobra Killuc Wildlife Reserve ....................................................................... 32  
  8) Mt Rothwell ................................................................................................... 35  
Synopsis .................................................................................................................. 36  

**Chapter 3: An assessment of three techniques for monitoring a rare, cryptic**  
grassland marsupial ............................................................................................. 43  
Abstract .................................................................................................................. 45  
Introduction ........................................................................................................... 46  
Methods ................................................................................................................. 47  
Results ..................................................................................................................... 52  
Discussion .............................................................................................................. 55  

**Chapter 4: Living with the enemy: dynamics of a declining population of the**  
critically endangered eastern barred bandicoot Perameles gunnii ................. 59  
Abstract .................................................................................................................. 61  
Introduction ........................................................................................................... 62
List of tables

Table 2.1: Summary of eastern barred bandicoot release sites from 1989 to 2005............. 37

Table 3.1: Eastern barred bandicoot monitoring data for Woodlands Historic Park from
winter 2006 to summer 2007. ........................................................................................................... 53

Table 3.2: Eastern barred bandicoot monitoring data for Hamilton Community Parklands
from winter 2007 to autumn 2009 ..................................................................................................... 54

Table 3.3: Output of the linear mixed model showing the seasonal estimated coefficients,
standard errors (SE) and back transformed coefficients, converted to a percentage, for all
significant variables collected from Hamilton Community Parklands between winter 2007
and autumn 2009............................................................................................................................... 55

Table 6.1: Output of the eastern barred bandicoot reintroduction phase (winter and spring
2007) and established phase (winter 2008 to autumn 2009) linear mixed models, showing
the estimated coefficients and standard errors (SE) for all significant habitat variables in
each season at Hamilton Community Parklands. ............................................................................... 104

Table 6.2: Descriptive statistics for founder eastern barred bandicoots caught in the
reintroduction phase (winter and spring 2007) and transition and established phases
combined (summer 2008 to spring 2009) at Hamilton Community Parklands. .................. 105

Table 6.3: Results from independent t-tests run in SPSS on 95% minimum convex polygons
(MCP), span, capture and plot data collected from founder eastern barred bandicoots
caught in the reintroduction phase (winter and spring 2007) and transition and established
phases combined (summer 2008 to spring 2009) at Hamilton Community Parklands. ...... 106

Table 6.4: Descriptive statistics, on untransformed data, for 95% minimum convex polygons
(MCP), span, number of times an individual eastern barred bandicoot was captured and the
number of different plots a bandicoot was captured at Hamilton Community Parklands from
winter 2007 to autumn 2009. ............................................................................................................. 106

Table 6.5: Results from general ANOVA of 95% minimum convex polygons (MCP), span,
captures and plots occupied by founder and wild-bred, and male and female eastern barred
bandicoots caught at Hamilton Community Parklands between winter 2007 and autumn
2009. ............................................................................................................................................. 107

Table 7.1: Means of foraging and nesting habitat variables at Mt Rothwell in autumn 2009
± standard error ................................................................................................................................. 123

Table 7.2: Generalised linear mixed model quantifying the effect of four predictor variables
(% lateral cover, >15 cm grass, litter and distance to cover) on the selection of locations for
foraging and nesting by eastern barred bandicoots at Mt Rothwell in autumn 2009. .... 124
Table 7.3: Summary of nest use by male and female eastern barred bandicoots at Mt Rothwell in autumn 2009.................................................................125

Table 7.4: Home range, range span and maximum distance travelled in 24 hours by male and female eastern barred bandicoots at Mt Rothwell in autumn 2009. The t-tests examine the differences between males and females within each variable. ..................................................127

Table 7.5: Home ranges calculated in previous studies for male and female eastern barred bandicoots using minimum convex polygons (MCP) or kernel contours (KL). .................130
List of figures

Fig 2.1: Former distribution of eastern barred bandicoots in Victoria and locations of all reintroduction sites...................................................................................................................................................... 14

Fig 2.2a: Number of eastern barred bandicoots captured at Woodlands Historic Park per 100 trap nights from August 1996 to April 2005. ........................................................................................................ 17

Fig 2.2b: Rainfall decile for Woodlands Historic Park from August 1996 to April 2005 .... 17

Fig 2.3a: Number of eastern barred bandicoots captured at Hamilton Community Parklands per 100 trap nights from March 1995 to June 2003.......................................................... 20

Fig 2.3b: Rainfall decile for Hamilton Community Parklands from March 1995 to June 2003. ....................................................................................................................................................................... 20

Fig 2.4a: Number of eastern barred bandicoots captured at Mooramong per 100 trap nights from January 1994 to July 2005. ........................................................................................................................................ 23

Fig 2.4b: Rainfall decile for Mooramong from January 1994 to July 2005. .................... 23

Fig 2.5a: Number of eastern barred bandicoots captured at Floating Islands Nature Reserve per 100 trap nights from November 1994 to August 1998. ............................................................ 25

Fig 2.5b: Rainfall decile for Floating Islands Nature Reserve from November 1994 to August 1998. .......................................................................................................................................................... 25

Fig 2.6a: Number of eastern barred bandicoots captured at Lake Goldsmith Wildlife Reserve per 100 trap nights from July 1996 to December 2003............................ 28

Fig 2.6b: Rainfall decile for Lake Goldsmith Wildlife Reserve from July 1996 to December 2003. ................................................................................................................................................................... 28

Fig 2.7a: Number of eastern barred bandicoots captured at Lanark per 100 trap nights from August 1996 to June 2004. .................................................................................................................... 31

Fig 2.7b: Rainfall decile for Lanark from August 1996 to June 2004. ............................ 31

Fig 2.8a: Number of eastern barred bandicoots captured at Cobra Killuc Wildlife Reserve per 100 trap nights from April 1997 to July 2001.. ................................................................. 34

Fig 2.8b: Rainfall decile for Cobra Killuc Wildlife Reserve from April 1997 to July 2001...... 34

Fig 3.1: Woodlands Historic Park, showing the 50 random plots and the location of the 99 plot grid used to trap, spotlight and count foraging digs of eastern barred bandicoots from winter 2006 to summer 2007. ........................................................................................................................................................................... 48
Fig 3.2: Hamilton Community Parklands, showing the location of the 60 plots used to trap, spotlight and count foraging digs of eastern barred bandicoots from winter 2007 to autumn 2009.

Fig 4.1: Mooramong, showing the location of the Nature Reserve, Cottage Dam and Homestead Gardens.

Fig 4.2: Population rate of increase of eastern barred bandicoots at Mooramong between consecutive trap sessions from January 1994 to March 2009.

Fig 4.3: Number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to March 2009.

Fig 4.4: Number of eastern barred bandicoots caught per 100 trap nights at Mooramong in the Nature Reserve, Cottage Dam and Homestead Gardens from February 1995 to October 2009.

Fig 4.5: Consecutive records of the six eastern barred bandicoots (sex and ID number) that moved between the Nature Reserve, Cottage Dam and Homestead Gardens at Mooramong from December 1995 to July 2005.

Fig 4.6: Cumulative frequency distribution chart showing disappearance rate of male and female eastern barred bandicoots at Mooramong from December 1992 to March 2009.

Fig 4.7: Linear regression of the population rate of increase on the number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to June 2008.

Fig 4.8: Linear regression of the proportion of females with pouch young on the number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to June 2008.

Fig 4.9: Linear regression of mean litter size on the number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to June 2008.

Fig 5.1: Population rate of increase of eastern barred bandicoots at Hamilton Community Parklands from October 2007 to Feb 2010.

Fig 5.2: Number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from July 2007 to February 2010.

Fig 5.3: Cumulative distribution frequency of the number of months male and female eastern barred bandicoots were known to be alive at Hamilton Community Parklands between July 2007 and February 2010.

Fig 5.4: Proportion of female eastern barred bandicoots with pouch young at Hamilton Community Parklands from July 2007 to February 2010.
Fig 5.5: Mean litter size of female eastern barred bandicoots at Hamilton Community Parklands from July 2007 to February 2010. ................................................................. 87

Fig 5.6: Linear regression of the population rate of increase on the number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from December 2007 to February 2010. ............................................................... 88

Fig 5.7: Linear regression of the proportion of females with pouch young on the number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from December 2007 to February 2010. .............................................................................................................. 89

Fig 5.8: Linear regression of mean litter size on the number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from December 2007 to February 2010. .............................................................................................................. 89

Fig 6.1: Hamilton Community Parklands showing the location of eastern barred bandicoot release sites in July 2007 and November 2007................................................................. 99

Fig 6.2: Mean number of eastern barred bandicoot digs (excluding zero values) in grassland and woodland areas from winter 2007 to autumn 2009 at Hamilton Community Parklands. .......................................................................................................................... 103

Fig 7.1: Mt Rothwell showing the areas of woodland (darker areas) and grassland (paler areas)................................................................................................................................. 116

Fig 7.2: Proportional habitat use at the home range and study area scale for foraging and nesting at Mt Rothwell in autumn 2009. ................................................................. 122

Fig 7.3: Kernel contours home range of a typical male (PIT tag # 0006C9754F) in this study, showing the location of foraging and nesting sites at Mt Rothwell in autumn 2009........ 122

Fig 7.4: Ordination diagram created using non-metric multidimensional scaling and a Euclidean resemblance matrix. Points on the diagram represent foraging and nesting locations at Mt Rothwell in autumn 2009. ................................................................. 123

Fig 7.5: Linear regression of the total number of nests known to be used by each eastern barred bandicoot (n = 11) at Mt Rothwell in autumn 2009 on the number of days radio-tracked. ................................................................................................................................. 126
Chapter 1

General introduction

Eastern barred bandicoot at Hamilton Community Parklands, October 2008 (photograph by Gavin Lewis)
Chapter 1

General introduction

Reintroduction biology

Reintroduction projects attempt to re-establish wildlife within their historical range through the release of wild or captive-bred individuals following extirpation or extinction in the wild (IUCN 1998). Many species are currently being driven to extinction by anthropogenic activities, which have resulted in habitat loss and alteration, overexploitation, introduced species, pollution and climate change (Mace et al. 2008). Griffith et al. (1989) foresaw that, with the increasing rate of species extinctions and impending reduction in overall biological diversity, reintroduction of rare species would become an increasingly important conservation technique. It is not clear when the first true reintroduction occurred, but it could have been the release of 15 American bison *Bison bison* into a newly established reserve in Oklahoma in 1907 (Seddon et al. 2007, Kleiman 1989). Despite this release being a success (Kleiman 1989), many subsequent reintroductions have failed because the vast majority of animals died soon after release (Teixeira et al. 2007). Many of these reintroductions involved the release of animals with little planning and often no monitoring (Seddon et al. 2007) and little was learnt from the process (Armstrong and Seddon 2008).

Despite reintroduction attempts occurring for at least 100 years, the field of reintroduction biology was born much later in response to the poor success of reintroduction programs (Armstrong and Seddon 2008). The term ‘reintroduction biology’ refers to research undertaken to improve the outcomes of reintroductions and other translocations carried out for conservation purposes (Armstrong and Seddon 2008). However, reintroduction research has generally been fragmented and ad hoc, rather than an organised attempt to gain the knowledge needed to improve the success of reintroductions (Seddon et al. 2007). Appropriate and well planned release programs can expand the scope of metapopulations of threatened species, as well as improve understanding of both the ecological requirements of a declining species and the factors that may threaten its survival in the wild (Serena and Williams 1995). However, the high costs associated with reintroduction projects make many reintroductions unfeasible as a conservation strategy for most rare and endangered species (Kleiman 1989, Serena and Williams 1995). Therefore cost effective monitoring and management approaches need to be developed and, as stressed by
Armstrong and Seddon (2008), monitoring needs to be designed to address questions identified *a priori*, which is a far more efficient use of conservation funds.

**Eastern barred bandicoots**

The Victorian eastern barred bandicoot *Perameles gunnii* is listed as Critically Endangered by the Advisory List of Threatened Vertebrate Fauna in Victoria 2007, Endangered under the *Environment Protection and Biodiversity Conservation Act* 1999, Threatened under the Victorian *Flora and Fauna Guarantee Act* 1988 and Near Threatened (Victorian and Tasmanian forms combined) on the IUCN Red List, due to the Tasmanian form being reasonably widespread and fairly common. This small (<1 kg) nocturnal marsupial belongs to the family Peramelidae and is cryptically coloured with three to four whitish bars across the rump (Backhouse and Crosthwaite 1996). It is solitary (Dufty 1991b), short lived (2 - 3 years), but highly fecund with a gestation of just 12.5 days (Backhouse *et al.* 1994). Females are capable of producing up to five litters throughout the year, with an average of two to three young in each litter (Backhouse *et al.* 1994). Young remain in the pouch for 53 days, are left in a nest and weaned by day 57, but remain dependent on their mother for some time (Dufty 1995). Females become sexually mature at 3 months and males at 5 months (Dufty 1995, Heinsohn 1966). Eastern barred bandicoots are omnivorous and opportunistically exploit a wide variety of soil and surface invertebrates, including crickets, beetles and worms, and are also known to eat plant matter such as bulbs of onion grass *Romulea rosea* and fallen fruit (Backhouse and Crosthwaite 1996, Dufty 1991b). Earthworms are the main food items during wet months, whilst insects become more common in the diet during the dry months (Heinsohn 1966).

At the time of European settlement in Australia the eastern barred bandicoot was found across the Basalt Plains of Victoria, extending from Melbourne to the South Australian border, an area of ~23,000 km² (Seebeck *et al.* 1990). The original vegetation of the Basalt Plains was perennial tussock grassland dominated by wallaby grasses *Danthonia* spp., spear grasses *Stipa* spp., tussock grass *Poa labillardieri*, and kangaroo grass *Themeda triandra* (Menkhorst and Seebeck 1990). Areas of savannah woodland were also present and were mostly composed of river red gum *Eucalyptus camaldulensis*, with some yellow box *E. melliodora*, swamp gum *E. ovata* and drooping sheoak *Casuarina stricta*, with a tussock
grassy understory (Menkhorst and Seebeck 1990). Climate, fire and predation, are believed to have been the major controlling influences on populations of eastern barred bandicoots prior to European settlement (Seebeck et al. 1990). Potential predators included the dingo Canis familiaris, spot-tailed quoll Dasyurus maculatus, eastern quoll D. viverrinus, nocturnal and diurnal birds of prey (e.g. wedge-tailed eagles Aquila audax) and snakes (e.g. tiger snake Notechis scutatus) (Seebeck et al. 1990).

By 1989 the population had contracted to ~600 ha in and around the city of Hamilton, which lies near the western margin of the Basalt Plains (Minta et al. 1990, Seebeck et al. 1990). At this time it was estimated that between 150 and 300 bandicoots remained (Lacy and Clark 1990). Habitat alteration due to farming practices and introduced predators (red fox Vulpes vulpes) and competitors (European rabbit Oryctolagus cuniculus) were thought to have caused this decline (Seebeck et al. 1990). In particular, predation by red foxes was thought to have been a significant contributor; eastern barred bandicoots were still abundant in Tasmania, which had also undergone significant habitat alteration but was free from red foxes (Seebeck et al. 1990). Today over 99% of Victoria’s native grasslands and grassy woodlands has been lost (Backhouse et al. 1994). In 1989 the first management plan for the eastern barred bandicoot was developed (Brown 1989). Since then, eight reintroduction sites have been established across the Basalt Plains, with varying success (Winnard and Coulson 2008). Only three sites now contain bandicoots: Hamilton Community Parklands, Mt Rothwell and Mooramong.

**Aims**

The main aim of this PhD was to conduct research that would improve the outcome of the eastern barred bandicoot recovery program, so that management techniques can be refined and population sizes increased. To do this I focussed on three main areas: finding the most cost and time effective monitoring techniques, determining the regulatory processes acting upon two very different populations (one with red foxes and one without), and identifying the habitat variables that are important for eastern barred bandicoot survival and persistence.
Thesis outline

Chapters in this thesis follow a journal article structure. Therefore there is no one chapter that outlines the study site, study species and common methods. New methods and relevant aspects of the study sites are described sequentially in each chapter as needed, and reference made to previous chapters where necessary. This structure inevitably results in some degree of overlap, particularly regarding study species, study sites and some methods.

In Chapter 2 I review the eastern barred bandicoot recovery program from when it first began in 1989 up to 2005, shortly before I began field work for this thesis. This chapter combines much grey literature and has been published in Pacific Conservation Biology. The current status of sites that still contain eastern barred bandicoots are given in subsequent chapters.

Chapter 3 determines the most time and cost effective monitoring techniques for a declining and an increasing population. Due to limited funds available to reserve managers, it is important that they use a technique that is both efficient in terms of time and money, and results in the best data collected that can be used to refine management actions.

Chapters 4 and 5 examine the regulatory processes in a declining population at Mooramong that is not fox free, and a recently reintroduced population at Hamilton Community Parklands that is protected from foxes by a predator-barrier fence. Chapter 4 concludes that, even though the Mooramong population has persisted for at least 17 years, its decline has largely been influenced by red fox predation. In contrast, in the fox-free Hamilton Community Parklands (Chapter 5), there is some evidence of an Allee effect.

Chapters 6 and 7 both examine the relationship between bandicoots and their habitat. Chapter 6 examines foraging habitat use at the population scale by a newly reintroduced population at Hamilton Community Parklands. The chapter concludes that the main predictor of foraging habitat is soil compaction, and that founder bandicoots occupy larger areas than wild-bred bandicoots. Chapter 7 was conducted at Mt Rothwell when the reserve had been severely overgrazed by macropods. This chapter reports that bandicoots were selecting the same types of habitat for foraging and nesting as found in other studies, but the distances they were travelling between the two were far greater.
Chapter 8 synthesises all chapters and makes management recommendations based on my findings. This chapter is intended primarily for reserve managers, and will aid in the development of more refined management techniques to achieve positive outcomes for the eastern barred bandicoot.
Chapter 2

Sixteen years of eastern barred bandicoot *Perameles gunnii* reintroductions in Victoria: a review


*Eastern barred bandicoot released at Hamilton Community Parklands, July 2007*
Chapter 2

Sixteen years of eastern barred bandicoot *Perameles gunnii* reintroductions in Victoria: a review

Abstract

Once widespread throughout Victoria, the eastern barred bandicoot *Perameles gunnii* has declined to near extinction on the Australian mainland due to habitat loss and predation by exotic predators. The last remaining wild population occurs in Hamilton, western Victoria. Founders for a captive breeding program were taken from this population in 1988, which has persisted without predator control or supplementation from captive-bred animals. The species was reintroduced to eight sites from 1989: Woodlands Historic Park, Hamilton Community Parklands, Mooramong, Floating Islands Nature Reserve, Lake Goldsmith Wildlife Reserve, Lanark, Cobra Killuc Wildlife Reserve and Mt Rothwell.

Although all reintroduced populations initially increased, declines were observed during the mid to late 1990s, from which they have never recovered. A combination of drought and predation by red foxes *Vulpes vulpes* and cats *Felis catus* is thought to be responsible. Currently, management techniques focus on intensive control of red foxes by poison baiting and shooting, and by construction and maintenance of predator-barrier fences at some sites.

Understanding which characteristics led to the success or failure of a reintroduction site is paramount to the success of the recovery program for this species. This paper reviews the status of all reintroduction sites, and examines characteristics that could account for the performance of their populations.
Introduction

The eastern barred bandicoot *Perameles gunnii* belongs to the family Peramelidae (Marsupialia). The species is short lived, with few surviving to three years in the wild, but is highly fecund: gestation lasts 12 days, young are weaned at 55 days, and females can produce five litters a year, with an average of two to three young in each (Backhouse *et al.* 1994). Sexual maturity is reached at 3 months for females and 5 months for males (Dufty 1995). Females can reproduce throughout the year, although reproduction is depressed in late summer and ceases during times of drought (Backhouse and Crosthwaite 1996).

Eastern barred bandicoots rest in a grass nest during the day, then emerge to forage within two hours after sunset (Dufty 1991b). They feed on a variety of invertebrates, including earthworms (Oligochaeta), beetles (Coleoptera) and crickets (Orthoptera), as well as fruit and bulbs of onion grass *Romulea rosea* (Backhouse and Crosthwaite 1996). Optimal habitat has been described by Dufty (1994b) as heterogeneous and structurally complex (i.e. areas with high floristic diversity, lack of grazing and low disturbance) with dense cover for nesting, adjacent to open areas for feeding (Watson and Hill 2005). Bandicoots have also been found to prefer areas with high soil moisture, such as along creek margins and swampy depressions (Watson and Hill 2005). A population density of 0.7 ha\(^{-1}\) is considered typical, and a maximum density of 8 ha\(^{-1}\) has been reported (Backhouse *et al.* 1994, Backhouse and Crosthwaite 1996). Home range size is 4.0 ha for males and 1.6 ha for females, with maximum range lengths of 260 m and 72 m for males and females respectively (Dufty 1991b, Dufty 1994b).

Prior to European settlement of Australia, the eastern barred bandicoot occurred in Tasmania, on the Basalt Plains of Victoria, and in a restricted area of south-eastern South Australia (Seebeck *et al.* 1990). The Basalt Plains were originally composed of perennial tussock grassland with patches of savannah woodland (Seebeck 1979). Dominant grass species were wallaby grasses *Danthonia* spp., spear grasses *Stipa* spp., tussock grass *Poa labillardieri* and kangaroo grass *Thymedt australis*, with the woodlands dominated by river red gum *Eucalyptus camaldulensis* (Seebeck 1979). Very little of these native grasslands remain, as indigenous species have been replaced by exotic pasture grasses, clovers and weeds (Seebeck 1979). This change in habitat initiated the decline of the eastern barred bandicoot (Seebeck 1979). Further declines in native grasses occurred due to heavy grazing of grasslands by domestic stock and the addition of fertilisers and insecticides (Seebeck...
1979). This altered the grassland flora and may also have decreased the density of soil invertebrates on which bandicoots depend for food (Seebeck 1979, Nkem et al. 2002).

Although the bandicoot is considered secure in Tasmania, this is not the case on the mainland (Seebeck et al. 1990). In South Australia the bandicoot is presumed extinct (Seebeck et al. 1990), and in Victoria only a few reintroduced populations remain, along with a remnant wild population along the Grange Burn (a creek) in Hamilton. Predation by introduced predators such as cats (*Felis catus*) (both feral and domestic) and the red fox (*Vulpes vulpes*) is thought to be one of the main threats to mainland populations (Krake and Halley 1993). This has led to captive-breeding of eastern barred bandicoots, which have been released into areas surrounded by a predator-barrier fence, or where regular red fox control can take place. Other possible threats to the current small, fragmented mainland populations include inadequate food and shelter due to stochastic events such as fire and drought.

The first management plan for the eastern barred bandicoot (Brown 1989) focused on the reintroduction of the species into Gellibrand Hill Park, near Melbourne, which was to be evaluated before reintroductions to other sites were considered. The plan listed the following site selection criteria: extensive suitable land with dense ground cover and open areas capable of supporting 50 breeding pairs, preferably managed by the then Department of Conservation, Forests and Lands or owned by adjoining private agricultural properties, and ongoing habitat management (Brown 1989). In 1993, criteria for success of released populations were set at 50% survival of individuals after one month and 10% survival after 6 months (Lawrence 1995). These criteria formed part of the eastern barred bandicoot reintroduction protocol for the third reintroduction, to Mooramong (Humphries and Seebeck 1995); they are now used to determine the success of all Victorian eastern barred bandicoot reintroductions.

In total, eight reintroduction sites have been established in Victoria (Fig 2.1), each with varying success. This review outlines the history of eastern barred bandicoot reintroductions at each site, evaluates possible reasons for their current status and recommends guiding principles for standardising monitoring across all reintroduction sites.
Reintroduction Sites

1) Woodlands Historic Park

Situated 22 km north of Melbourne, Woodlands Historic Park, formerly Gellibrand Hill Park, was the site of the first eastern barred bandicoot reintroduction in 1989. Woodlands Historic Park was considered an ideal reserve for bandicoots, as it was managed by the Victorian Department of Conservation and Environment (Reading et al. 1991), which meant full time staff and facilities could be provided (Watson 1991). The presence of captive breeding facilities and the location of the reserve within the bandicoots former range were also taken into consideration (Reading et al. 1991). Once chosen as a release site, a 1.8 m high predator-barrier fence (Backhouse et al. 1997) was constructed around 400 ha of open grassy woodland (Seebeck 1990). This area was later reduced to 300 ha, excluding a weak point where the fence crossed a creek. Public access into the reserve was, and still is permitted.
Initially Woodlands Historic Park was used as an eastern barred bandicoot captive breeding site. Animals were collected in July 1988 from the last remaining wild population at Hamilton and released (Seebeck 1990) into 100 - 200 m² pens. The original design of the pens permitted free movement of juveniles into the reserve while inhibiting adults (Reading et al. 1991). This design was later modified to inhibit all movement, with the aim of monitoring breeding success (Reading et al. 1991), but this was never quantifiable (Michael Lynch pers. comm.). In late 1991, the breeding pens were abandoned due to injuries caused by the fence, mortality sustained through fighting, a black rat *Rattus rattus* problem due to supplementary feeding, and the inability to monitor the number and health of bandicoots (Michael Lynch pers. comm.). Captive breeding was then moved into properties managed by the Zoological Board of Victoria, and an intensive breeding program began.

Prior to releases into the reserve, extensive predator control was carried out by shooting, poisoning with 1080 meat baits and snare trapping (Seebeck 1990). With the maturation of breeding stock and a satisfactory level of red fox control, a trial reintroduction occurred in April 1989 (Seebeck 1990). Of the ten bandicoots released, only five were re-captured: three others were killed by a canid, another died of unknown causes and the remaining one was never seen again (Seebeck 1990). A greater intensity of predator control was deemed necessary before any subsequent releases (Seebeck 1990).

A total of 62 bandicoots (30 male and 32 female) were released into the reserve between April 1989 and July 1991 (Murphy 1991). Of these, 37% were known to be alive, 15% were found dead and 48% were missing (not seen in three months) in July 1991 (Murphy 1991). Pouch young were observed in 70% of females trapped in April 1991 (Murphy 1991). An estimated population of 90 bandicoots was present in 1992, prior to the release of a further 24 individuals (Backhouse et al. 1994). Drainage lines and areas of highest vegetative cover were prime habitat, as determined by trap success and habitat assessments at each trap station (Seebeck et al. 1993). In an attempt to extend and enhance the habitat, grazing pressure was reduced by the removal of around 40,000 - 50,000 European rabbits *Oryctolagus cuniculus* in 1993 (Seebeck et al. 1993). The population of bandicoots reached its peak in 1994/95 with over 600 believed to be occupying the entire reserve (Seebeck 1999). The success of Woodlands Historic Park at this stage was attributed to the control of introduced predators and European rabbits, and a series of high rainfall seasons (Seebeck 1999) that enhanced soil moisture and invertebrate content.
A population decline began in late 1995 (Seebeck 1999). Predation was not considered a major factor, but rather the dry conditions in the reserve coupled with an increasing population of European rabbits and eastern grey kangaroos *Macropus giganteus*, which reduced available habitat (Seebeck 1999). Another contributing factor to the decline was the removal of at least 100 bandicoots between 1995 and 1997 to supplement populations at other release sites (Backhouse et al. 1997). Delayed maturation and a reduction in breeding were also recorded at this time (Seebeck 1999). The population declined rapidly and only two bandicoots were trapped in 1998 (Fig 2.2a) (Long et al. 2004); these individuals were placed in captivity and the population at Woodlands Historic Park was considered extinct (Long et al. 2004). A program to control European rabbits and eastern grey kangaroos began in the reserve (Seebeck 1999). Bandicoot diggings were subsequently observed (Long et al. 2004) and two small releases of eight and ten bandicoots occurred in 1999 and 2001 respectively. The population remains small, with most trap sessions returning zero individuals (Fig 2.2a).

Current management problems associated with the site include visitor behaviour, predator control, and low rainfall. Members of the public leave gates open or vandalise the predator-barrier fence allowing red foxes easy access. Upgrades to the fence have recently been made but a ‘floppy top’ is still required. Once installed, this, along with regular fence maintenance, should result in red foxes being absent for the majority of the time. Eastern grey kangaroo and European rabbit numbers continue to be controlled, which has a positive effect on the amount of cover available despite the number of below average rainfall years. The current management plan highlights the need to improve predator control and manage site conditions to maximise survival of reintroduced animals (Watson and Hill 2005).
a) Number of eastern barred bandicoots captured at Woodlands Historic Park per 100 trap nights from August 1996 to April 2005. Data from 1989 to 1995 is not shown, due to insufficient data recorded on the number of traps set. Founders: chipped bandicoots released from captivity or other sites. Recruitment: any unchipped bandicoot born at that site. Wild-bred: chipped bandicoot born at that site.

b) Rainfall decile for Woodlands Historic Park from August 1996 to April 2005. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Melbourne airport (86282) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; $y = -0.0865 x +5.7875, r^2 = 0.0003, P = 0.28$
2) Hamilton Community Parklands

The city of Hamilton is situated 280 km west of Melbourne and supports the last wild population of eastern barred bandicoots (Richard Hill pers. comm.). This presumably small population is restricted to vegetation along the Grange Burn. As this population continued to decline through the 1980s, the Hamilton Institute of Rural Learning, along with strong community support, initiated the establishment of a release site within the Hamilton Community Parklands (Backhouse et al. 1994). A 100 ha area previously used for grazing and recreation was enclosed by a 1.8 m high electrified fence, completed in 1989 (Backhouse et al. 1994). This site was chosen due to the presence of native grasslands, tree plantings and a permanent water supply (Peter Goldstraw pers. comm.). One small breeding pen (~30 × 15 m) was incorporated into the fenced area (Seebeck 1990), but the number of bandicoots born in the pen is unknown, as juvenile movement into the reserve was not restricted (Peter Goldstraw pers. comm.).

Captive breeding and reintroduction of eastern barred bandicoots into Hamilton Community Parklands commenced shortly after the Woodlands Historic Park program in 1989 (Reading et al. 1991). This action deviated from the original management plan (Brown 1989), due to a need to increase the captive and semi-captive population rapidly, the presence of optimal bandicoot habitat, and strong community support for a bandicoot release (Reading et al. 1991).

Eastern barred bandicoots in the first release were retained in artificial hard shelters scattered throughout the reserve (Reading et al. 1991). After several hours the animals were allowed access to the reserve; supplementary feeding occurred for six to ten days (Reading et al. 1991). Annual population monitoring began in 1990, with 20 individuals trapped (Reading et al. 1991). The population gradually increased, and by 1993 Hamilton Community Parklands had approximately 90 bandicoots (Backhouse et al. 1997), but this population size was short-lived. Monitoring during 1994 indicated a significant decline (Backhouse et al. 1997) possibly due to deficiencies in the fence design, and a long period of drought (Peter Goldstraw pers. comm.). Poisoning and shooting of red foxes increased immediately before supplementing the population with 28 animals the following year (Backhouse et al. 1997). Bandicoot numbers increased, and the population appeared to stabilise, with some seasonal fluctuations (Watson 1997b). A second decline occurred in late 1997 (Fig 2.3a); this decline continued and monitoring in August 1999 resulted in zero.
captures. The population recovered and appeared to stabilise with seasonal fluctuations, before being supplemented with a release of 16 bandicoots. This release made no impact on population size and 2004 was the start of a series of monitoring sessions that resulted in zero captures.

A dry period began in Hamilton in 1997, but there are several factors that could have contributed to the second decline. Hamilton Community Parklands, like Woodlands Historic Park, is a public access reserve. Problems associated with public access are similar at the two sites, but Hamilton Community Parklands has also suffered from the deliberate release of domestic cats (Backhouse et al. 1997). The fence design was also ineffective at excluding red foxes, so an upgrade was completed in July 2005 by adding a ‘floppy top’ and running the hot wires off mains power instead of solar power (Peter Goldstraw pers. comm.). Intensive predator control has continued and no red foxes have been observed within the reserve since the ‘floppy top’ was installed (Peter Goldstraw pers. comm.). It is currently thought that no eastern barred bandicoots are in the Hamilton Community Parklands, but a release is planned for autumn 2007.
Fig 2.3a: Number of eastern barred bandicoots captured at Hamilton Community Parklands per 100 trap nights from March 1995 to June 2003. Data from 1989 to 1994 is not shown, due to insufficient data recorded on the number of traps set. Definitions as per Fig 2.2a.

Fig 2.3b: Rainfall decile for Hamilton Community Parklands from March 1995 to June 2003. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Hamilton airport (90173) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; 

\[ y = -0.0361x + 2.7569, \quad r^2 = 0.0026, \quad P < 0.01. \]

*No data.
3) Mooramong

Mooramong lies in the western district of Victoria 170 km west of Melbourne. This 1,562 ha working sheep farm is managed by the National Trust of Australia (Victoria) (Humphries and Seebeck 1995). A nature reserve has been established about 400 m from the Homestead in 243 ha of existing ephemeral wetlands (Humphries and Seebeck 1995), of which 113 ha is a mixture of grassland and shrubland suitable for eastern barred bandicoots (Backhouse et al. 1997). The majority of the reserve is dominated by introduced pasture grasses such as canary grass *Phalaris aquatica* and Yorkshire fog *Holcus lanatus* (Humphries and Seebeck 1995).

At the time plans were being made for releases into Woodlands Historic Park and Hamilton Community Parklands, the National Trust offered Mooramong as the third release site (Humphries and Seebeck 1995). Tall dense vegetation close to short pasture and wetland fringes made Mooramong apparently ideal for eastern barred bandicoots (Humphries and Seebeck 1995). Predator control was a concern at Mooramong, as there was no protective barrier against red foxes around the reserve (Backhouse et al. 1994). Community involvement in predator control and support for the program was sought through open days, publications and guided tours (Humphries and Seebeck 1995). However, local landholders believed there was no need to control red foxes, either because they farmed cattle, and red foxes did not pose a problem, or because they thought that red fox numbers were low on their properties (Jim O’Brien pers. comm.).

As with Woodlands Historic Park and Hamilton Community Parklands, a captive breeding colony was first established in the reserve (Humphries and Seebeck 1995). The first deliberate release occurred in 1992, when the Victorian captive population was saturated (Humphries and Seebeck 1995). The captive colony at Mooramong was released into the reserve and the breeding pens then became acclimatisation pens for bandicoots bred at other facilities (Backhouse et al. 1994). This allowed for individuals to adapt to local conditions, as well as allowing body condition and weight to be closely monitored prior to release (Backhouse et al. 1994).

The first release of 15 individuals occurred in December 1992 (Humphries and Seebeck 1995); 85 bandicoots were released over the next 18 months (Backhouse et al. 1997). Pouch young were recorded within one month of release, and the first wild-bred bandicoot was captured three months later (Humphries and Seebeck 1995). By the end of the first
year, wild-bred bandicoots had been recorded with pouch young (Backhouse et al. 1997) and constituted 76% of the population in April 1994 (Humphries and Seebeck 1995). In 1998/99 a decline in numbers trapped was recorded (Fig 2.4a) and attributed to the increasingly dry conditions.

During this time, many bandicoots shifted from the reserve to the Homestead Gardens where the lawns and garden beds are watered regularly (Jim O’Brien pers. comm.). Breeding continued and the population quickly recovered without any help from captive releases (Fig 2.4a). After two years of continued trapping in the reserve with no success, a decision was made to focus the monitoring effort on the Homestead Gardens (Jim O’Brien pers. comm.). Eastern barred bandicoots are now found around the Homestead in relatively high numbers, but digs have recently been observed in the reserve (O’Brien 2005). Mooramong is currently believed to support the largest population of eastern barred bandicoots in Victoria.
Fig 2.4a: Number of eastern barred bandicoots captured at Mooramong per 100 trap nights from January 1994 to July 2005. Data from 1992 to 1993 is not shown, due to insufficient data recorded on the number of traps set. Definitions as per Fig 2.2a.

Fig 2.4b: Rainfall decile for Mooramong from January 1994 to July 2005. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Skipton Post Office (89025) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; $y = 0.2426 \times +7.3196$, $r^2 = 0.0183$, $P < 0.01$. 
4) Floating Islands Nature Reserve

Situated 150 km southwest of Melbourne, Floating Islands is an 85 ha reserve. A mixture of ephemeral and permanent wetlands surrounds a mosaic of open grasslands, dense shrublands and grassy woodlands (Backhouse et al. 1997, Cook 2001). The terrain is rocky and steep as it occurs within basaltic stony rises (Cook 2001). Floating Islands was chosen as a release site due to the availability of suitable cover and water surrounded by private land (Steve McDougall pers. comm.). At the time of the first release, the site was managed by the Department of Natural Resources and Environment (formerly the Department of Conservation and Natural Resources) (Steve McDougall pers. comm.).

The first release of 22 bandicoots occurred in September 1994 (Backhouse et al. 1997). Pouch young were observed three months later (Backhouse et al. 1997), and a wild-bred sub-adult was trapped six months after the first release (Fig 2.5a) (Backhouse and Slater 1995). By August 1996 a total of 50 bandicoots had been released, and pouch young continued to be recorded throughout 1996 (Watson 1997b), but only two other wild-bred bandicoots were ever recorded (Steve McDougall pers. comm.).

Predator control was carried out with ‘Foxoff’ 1080 poisoning by a contractor (Watson 1997b) and neighbouring landholders actively contributed to predator control on their properties (Backhouse et al. 1997). In 1996/97 Parks Victoria took over management of the reserve (Steve McDougall pers. comm.), but maintaining adequate red fox control proved too difficult in the reserve (Watson and Hill 2005). Surrounding properties also reduced their red fox control effort due to animosity between landholders and Parks Victoria (Steve McDougall pers. comm.). The last poisoning to take place on the reserve was in 1998 (Steve McDougall pers. comm.).

Trapping showed a decline since the most successful monitoring session in December 1995 (Fig 2.5a). However, regular monitoring was difficult due to the rugged nature of the terrain (Backhouse et al. 1997). The population was presumed extinct after two unsuccessful trap sessions (Watson and Hill 2005). The failure of Floating Islands as a release site was attributed to the difficulties in maintaining red fox control (Steve McDougall pers. comm.).
Chapter 2: Reintroductions

**Fig 2.5a:** Number of eastern barred bandicoots captured at Floating Islands Nature Reserve per 100 trap nights from November 1994 to August 1998. One wild-bred female caught in 1999 is not shown, due to insufficient data on the number of traps set. Definitions as per Fig 2.2a.

**Fig 2.5b:** Rainfall decile for Floating Islands Nature Reserve from November 1994 to August 1998. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Colac Shire office (90147) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; \( y = 0.3569 x + 1.507, R^2 = 0.2701, P = 0.17. \) * No data.
5) Lake Goldsmith Wildlife Reserve

Lake Goldsmith Wildlife Reserve (870 ha) is located 50 km west of Ballarat and is now managed by Parks Victoria. A shallow temporary lake occupies the majority of the reserve in years of high rainfall (Chandler 2001). Of the 150 ha of dry land, 100 ha constitutes an isthmus which projects into the lake (Backhouse et al. 1997). The isthmus at Lake Goldsmith was chosen as an eastern barred bandicoot release site for a number of reasons. When the lake is full, 80% of the isthmus is surrounded by water, creating a partial barrier to predators (Chandler 2001). The isthmus also supports adequate native vegetation. In addition, a link was envisaged between Lake Goldsmith and Mooramong (15 km south of Lake Goldsmith) via revegetated drainage systems and nodes of habitat on private land (Humphries and Seebeck 1995).

Prior to release, a rigorous red fox control program was implemented by the Department of Conservation and Natural Resources in conjunction with adjacent landholders (Lawrence 1995). Methods included poison baiting with ‘Foxoff’, den destruction, organised shoots, and baiting large cat traps with European rabbit (Lawrence 1995).

The first release of nine bandicoots occurred in October 1994 (Backhouse et al. 1997), and pouch young were recorded two months later (Lawrence 1995). Even though breeding appeared successful, the first wild-bred bandicoot was not trapped until a year after the first release (Lawrence 1995). Of the 33 released in 1994, only eight bandicoots were encountered again, one of which died of predation (Lawrence 1995). A gradual decline in numbers captured was observed, but the success criteria were achieved (Lawrence 1995).

From 1997 the lake was predominantly dry (Chandler 2001), and throughout 1998 there was a decline in bandicoots captured (Fig 2.6a). During this time the red fox control program was intensified, with four poisoning programs every year (Chandler 2001). Further releases occurred late in 1998 to supplement the population. A few wild-bred individuals were captured up to December 2000, but the individuals trapped were predominantly 1998 founders. After an initial increase, numbers declined again; trapping in October 2001 detected no bandicoots (Fig 2.6a). A small release occurred in late 2001 but the population remained low. Breeding at Lake Goldsmith continued, as was evident from two untagged individuals captured in March 2004 (Hall 2005). Since then conditions at Lake Goldsmith have improved due to higher rainfall but bandicoot captures have remained low, with only one female captured in January 2005 (Hall 2005). All active management has been
suspended at Lake Goldsmith, and a decision on the site’s role in the recovery effort will be made after studies have determined the potential of this site to support a viable population of eastern barred bandicoots (Richard Hill pers. comm.).
**Fig 2.6a:** Number of eastern barred bandicoots captured at Lake Goldsmith Wildlife Reserve per 100 trap nights from July 1996 to December 2003. Data from October 1992 to July 1996 is not shown, due to insufficient data recorded on the number of traps set. A female captured in Jan 2005 is also not shown, due to her Trovan PIT (passive integrated transponder) tag number being recorded incorrectly and therefore it is not known whether she is a founder or a wild-bred individual. Definitions as per Fig 2.2a.

**Fig 2.6b:** Rainfall decile for Lake Goldsmith Wildlife Reserve from July 1996 to December 2003. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Beaufort (89005) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; $y = 0.0535x + 2.9409$, $r^2 = 0.0041$, $P < 0.01$. 

![Figure 2.6a](image-url)  
![Figure 2.6b](image-url)
6) Lanark

Lanark is an 800 ha diversified farming property situated 340 km west of Melbourne (Backhouse et al. 1997). The property contains 63 ha of permanent and ephemeral wetlands, and 48 ha of revegetated shelterbelts and bush blocks (Backhouse et al. 1997). These met a number of criteria for a suitable release site (Peter Goldstraw, pers. comm.). The remaining area consists of open sheep paddocks, and stands of timber for commercial tree farming (Backhouse et al. 1997).

Extensive predator control was carried out on the property (Watson 1997b); baiting with ‘Foxoff’ occurred weekly to fortnightly throughout the year, along with spotlight shooting prior to and during lambing seasons (Watson 1997b). Adjoining properties were encouraged to take up the baiting program and field days were held to promote red fox control and outline the eastern barred bandicoot reintroduction program (Watson 1997b).

The first release of 10 bandicoots occurred in November 1994 and numerous small releases followed, with a total of 55 individuals being released up to November 1996. Evidence of breeding was observed in April 1995, but recruitment was not recorded until November 1996 (Fig 2.6a) (Watson 1997b).

In 1996, the wetlands began to dry, particularly the main lake, which contracted to the original small dam (Peter Goldstraw pers. comm.). Despite the contraction of the wetlands, the bandicoot population continued to increase to a peak in 1998/99 (Fig 2.7a). The population dispersed throughout the reserve and occupied the Homestead Gardens (Backhouse et al. 1997). At each monitoring session in 1998/99 at least 12 bandicoots were captured, no less than 30% being wild-bred and never encountered before (Peter Goldstraw pers. comm.). As the water receded, the prolific plant growth declined and the drying soil presumably decreased the availability of invertebrates. The population gradually declined from March 2000 despite two small releases in September 2001 and March 2002; monitoring in March 2003 recorded zero bandicoot captures (Fig 2.7a). Since then, one bandicoot was captured in June 2003 and one was observed in the Homestead Gardens 18 months later (Peter Goldstraw pers. comm.). In January 2004, the property was left vacant for two to three months. A reduction in predator control occurred, particularly the poison baiting program, but some spotlighting and shooting continued (Peter Goldstraw, pers. comm.). No bandicoots or their diggings were observed in 2005 (Peter Goldstraw, pers. comm.). All active management and predator control has been suspended at Lanark until
studies have determined the potential of this site to support a viable population of eastern barred bandicoots (Richard Hill pers. comm.).
Fig 2.7a: Number of eastern barred bandicoots captured at Lanark per 100 trap nights from August 1996 to June 2004. Data from 1992 to August 1996 is not shown, due to insufficient data recorded on the number of traps set. Definitions as per Fig 2.2a.

Fig 2.7b: Rainfall decile for Lanark from August 1996 to June 2004. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Branxholme (Basset) (90010) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; \( y = -0.0662 \times +5.2914, r^2 = 0.0026, P < 0.01. \)
7) Cobra Killuc Wildlife Reserve

Cobra Killuc is located 250 km west of Melbourne. It is a 500 ha public land reserve containing a variety of vegetation types, including remnant native grasslands and grassy woodlands, with plantations of native and exotic trees (Backhouse et al. 1997). Water bodies on site comprise a few artificial dams.

Cobra Killuc was chosen as the seventh reintroduction site as the available habitat met some of the selection criteria (Peter Goldstraw pers. comm.). Prior to any releases, extensive predator control was carried out using a combination of poisoning with 1080 and cyanide, shooting, and hunting with dogs (Watson 1997b). Adjoining landholders also participated in the ‘Foxoff’ program (Watson 1997b). An estimated 150,000 ha of land was involved in red fox control (Watson 1997b), but the number of baits laid and the frequency of baiting across this area is unknown.

The first release of 30 bandicoots occurred in April 1997. Six weeks later a monitoring session was carried out, and three females carrying a total of nine pouch young were trapped (Watson 1997b). A second release of 21 bandicoots occurred in September 1997; the first and only wild-bred bandicoot recorded at Cobra Killuc was captured two months later (Fig 2.8a) (Watson 1997a). By this time the number of captures had declined. Subsequent releases occurred in April 1998 (32 released), December 1998 (10 released) and May 1999 (10 released).

All eastern barred bandicoots released in December had radio-tags attached, which revealed a problem with red fox predation (Michael Lynch pers. comm.): over half of the bandicoots were killed within four days (Howard and Bedford 1999). Red fox control was increased before another 10 animals were released with radio-tags in May 1999, but red foxes also preyed heavily on these bandicoots (Michael Lynch pers. comm.). As a result, the decision was made to trap all the remaining bandicoots and return them to captivity (Howard and Bedford 1999). Monitoring continued, and October 2000 was the first monitoring session when zero bandicoots were captured (Fig 2.8a). Since then only one has been captured, in January 2001. Eastern barred bandicoot digs have not been observed since 2001 and the population is now considered extinct.

Although Cobra Killuc had initial breeding success, the site experienced management problems. Soon after the first bandicoot release, Parks Victoria took over management of
the site. Shortly after, a reduction in community involvement was evident, and predator control declined in the reserve and surrounding properties (Peter Goldstraw pers. comm.). Cobra Killuc was also affected by drought, which began in 1996. All available water bodies dried up, and invertebrate life and grass cover declined (Peter Goldstraw pers. comm.).
Fig 2.8a: Number of eastern barred bandicoots captured at Cobra Killuc Wildlife Reserve per 100 trap nights from April 1997 to July 2001. Definitions as per Fig 2.2a.

Fig 2.8b: Rainfall decile for Cobra Killuc Wildlife Reserve from April 1997 to July 2001. Data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Caramut Post Office (90136) station. Regression was calculated for the number of bandicoots captured per hundred trap nights and the rainfall decile for the previous quarter; \( y = -0.1378x + 1.9716; r^2 = 0.0857; P < 0.01. \)
8) Mt Rothwell

Mt Rothwell is situated at the northern end of the You Yangs Ranges, 60 km west of Melbourne. The 1,185 ha property was acquired by Earth Sanctuaries Ltd (now ES Link Pty Ltd) in March 2000 (Doherty 2002). Up to 70% of the native vegetation in the area was originally cleared for sheep grazing and cereal cropping. Plans are now in place to revegetate the whole area with native species (Doherty 2002). In December 2000, construction of a 5 km predator-barrier fence was completed. An intensive predator shooting, poisoning and den fumigation program followed and continued through 2001/02. In addition, about 1,200 European rabbit warrens were destroyed and 500 European rabbits were shot and poisoned during this period.

Although the 170 ha of open grassy woodlands offered ideal eastern barred bandicoot habitat, and the reserve was free of red foxes, the Recovery Team was concerned about plans to release eastern quolls *Dasyurus viverrinus* into the same area (Joel Little pers. comm.). However, Earth Sanctuaries Limited assured the Recovery Team that eastern quolls would have no impact on the bandicoots, since the two species coincided in Tasmania and no evidence of predation had been recorded (Joel Little pers. comm.).

In autumn 2004, the first release occurred into a 170 ha area of grassy woodlands (Joel Little pers. comm.); four animals were soft released into a 1 ha pen. A hard release occurred in December 2004 adjacent to the pens (Joel Little pers. comm.). These sites were chosen due to their high structural complexity and within two weeks of release, the hard release bandicoots had moved to open grassland nearby (Joel Little pers. comm.).

Monitoring in June 2005 resulted in capture of the first wild-bred eastern barred bandicoot at Mt Rothwell, two more releases of six animals have since occurred. The population appears to be increasing, with continued observations of pouch young and wild-bred individuals, but it is too early to measure success of the site. There have been incidents when the fence was vandalised or damaged by storms, but no red fox incursions have occurred. Red fox control programs are continuing, as is the expansion of native habitat. The habitat available to bandicoots has recently been expanded to include the adjacent grassland paddock (415 ha). This paddock is also surrounded by a predator-barrier fence and is considered to be free of introduced predators.
Synopsis

The reintroduction of eastern barred bandicoots has not been as successful as first anticipated. Identifying the exact cause of the failure of a site or the decline in a population is not possible in retrospect, due to a lack of detailed investigations or monitoring at the time. Table 2.1 summarises all available data. This section aims to provide a clearer understanding of the possible threatening processes influencing eastern barred bandicoot populations, allowing management techniques to be refined.

Drought

The importance of water to eastern barred bandicoots is not fully understood. Prolonged dry conditions would be likely to result in a reduction in cover and invertebrates, and both are considered important to bandicoots. Seebeck (1979) reported that the wild eastern barred bandicoot population at Hamilton declined markedly during the drought years of 1966 - 68, contracting to areas of permanent springs and streams. This movement to moist refugia was attributed to the greater accessibility of food in moist soils (Seebeck et al. 1990). Low rainfall has also been found to have an effect on eastern barred bandicoots in Tasmania, where Mallick et al. (2000) reported a decline in bandicoots during years of exceptionally low rainfall.

Drought can have a number of effects on wildlife, of which the most extreme is death, due to reduced food availability and/or quality (Cairns and Grigg 1993, Robertson 1986). Other consequences of drought include a reduction in body mass and reproduction (Lawrence et al. 2004), and biased sex ratios (Fisher 1999). Increased dry conditions may also reduce the availability of soil macroinvertebrates (Plum and Filser 2005). Drought can also induce predation on a declining prey population by reducing food and cover, and reducing the abundance of alternate prey for predators (Catling 1988). This effect could apply to eastern barred bandicoot populations: a decline in food and shelter for bandicoots would force them to forage longer each night and in more exposed habitat, making them more susceptible to predation. If the European rabbit population also declined due to insufficient food, red foxes could switch to alternate prey (i.e. bandicoots) as their primary prey, the European rabbit, declined. Prey switching has been reported in the red fox (Kjellander and Nordstrom 2003) and increased predation on native animals observed after European rabbit control (Haselmayer and Jamieson 2001).
### Table 1: Summary of eastern barred bandicoot release sites from 1989 to 2005

<table>
<thead>
<tr>
<th>Site</th>
<th>Size (ha)</th>
<th>Currently managed by</th>
<th>Predator-barrier fence</th>
<th>Captive breeding site</th>
<th>Habitat</th>
<th>Community involvement with fox control</th>
<th>Releases</th>
<th>No. EBBs in first release</th>
<th>Total no. of EBBs released</th>
<th>Possible reasons for decline</th>
<th>Current status of site</th>
</tr>
</thead>
<tbody>
<tr>
<td>Woodlands Historic Park</td>
<td>300</td>
<td>Parks Victoria</td>
<td>Yes</td>
<td>Yes</td>
<td>Mixed grassland, savannah woodland</td>
<td>No</td>
<td>Apr 89 – Apr 04</td>
<td>10</td>
<td>174*</td>
<td>Drought, overgrazing, removal of EBBs for translocation</td>
<td>Continuing</td>
</tr>
<tr>
<td>Hamilton Community Parklands</td>
<td>100</td>
<td>Southern Grampians Shire</td>
<td>Yes</td>
<td>Yes</td>
<td>Native grassland and grassy woodland</td>
<td>Yes</td>
<td>May 89 – Mar 03*</td>
<td>1*</td>
<td>122</td>
<td>Drought, issues with predator control</td>
<td>Continuing</td>
</tr>
<tr>
<td>Mooramong</td>
<td>200</td>
<td>National Trust</td>
<td>No</td>
<td>Yes</td>
<td>Open grasslands, permanent and ephemeral wetlands</td>
<td>No</td>
<td>Dec 92 – May 95*</td>
<td>15</td>
<td>85*</td>
<td>Drought</td>
<td>Continuing</td>
</tr>
<tr>
<td>Floating Islands</td>
<td>85</td>
<td>Parks Victoria</td>
<td>No</td>
<td>No</td>
<td>Open grasslands, dense shrublands and woodlands, swampy depressions</td>
<td>Yes, initially</td>
<td>Sep 94 – Aug 96</td>
<td>22</td>
<td>50</td>
<td>Management issues, difficulties in predator control</td>
<td>Considered extinct in 1999</td>
</tr>
<tr>
<td>Lake Goldsmith</td>
<td>150</td>
<td>Parks Victoria</td>
<td>No</td>
<td>No</td>
<td>Open grassy woodland and ephemeral wetlands</td>
<td>Encouraged</td>
<td>Oct 94 – Nov 01</td>
<td>10</td>
<td>63*</td>
<td>Most likely drought</td>
<td>Management suspended</td>
</tr>
<tr>
<td>Lanark</td>
<td>111</td>
<td>Privately owned</td>
<td>No</td>
<td>No</td>
<td>Open grasslands, woodland, permanent and ephemeral wetlands</td>
<td>Yes</td>
<td>Nov 94 – Mar 02*</td>
<td>10</td>
<td>63*</td>
<td>Drought and temporary reduction in predator control</td>
<td>Management suspended</td>
</tr>
<tr>
<td>Cobra Killuc</td>
<td>500</td>
<td>DSE</td>
<td>No</td>
<td>No</td>
<td>Remnant native grasslands, grassy woodlands</td>
<td>Yes, initially</td>
<td>Apr 97 – May 99</td>
<td>30</td>
<td>103</td>
<td>Difficulties in predator control</td>
<td>Considered extinct in 2002</td>
</tr>
<tr>
<td>Mt Rothwell</td>
<td>585</td>
<td>ES Link Pty Ltd</td>
<td>Yes</td>
<td>No</td>
<td>Open grassy woodland and grasslands</td>
<td>No</td>
<td>Oct 04 – Nov 05</td>
<td>4</td>
<td>22</td>
<td>n/a</td>
<td>Continuing</td>
</tr>
</tbody>
</table>

* Records unclear; EBBs = eastern barred bandicoots, DSE = Department of Sustainability and Environment
All sites except Floating Islands reported dry conditions from 1996 onwards, and declines in eastern barred bandicoot captures at this time have often been attributed to these conditions. The monthly rainfall decile at each release site for the corresponding trapping month is shown in Figs 2.2b - 8b. Regression analysis showed no significant relationship between bandicoot numbers and rainfall decile for the quarter prior to trapping at any site. However, the difficulties in trapping eastern barred bandicoots have made robust population estimates impossible to calculate.

The effects of low rainfall may take several high rainfall years to reverse, as suggested by Mallick et al. (2000) for Tasmanian eastern barred bandicoots. For example, Woodlands Historic Park experienced low rainfall in 1997 and the population has remained low, while rainfall has increased. It is unlikely that drought is entirely to blame for the low population density currently at Woodlands Historic Park, but it may have been a trigger for the decline.

Mooramong experienced a small population decline in 1998/99, from which the population recovered relatively quickly. This decline has been attributed to dry conditions experienced at Mooramong, but the regression analysis did not support this interpretation. However, it is possible that the apparent movement of the bandicoot population from the reserve to the Homestead Gardens, which are watered regularly, was a response to these conditions. During dry conditions, the gardens probably offered adequate shelter from predators as well as abundant invertebrates.

**Management**

Red foxes play a major part in reducing eastern barred bandicoot populations, as is evident from the many individuals found killed after reintroductions. Other reintroductions of endangered Australian fauna have experienced similar problems. In a reintroduction of burrowing bettongs *Bettonia lesueur* into Heirisson Prong, Western Australia, 36 - 77% of the population was killed in each of three red fox incursions through a predator-barrier fence (Short et al. 2002). Similarly, a reintroduction of 700 quokkas *Setonix brachyurus* into a fenced reserve south of Perth, Western Australia, over 12 years was unsuccessful because attempts to exclude red foxes and feral cats were ineffective (Short et al. 2002). Survival of prey species can be increased in several ways. One approach could be to modify habitat by the creation of predator refuge. For example, Ylönen et al. (2003) found that field voles *Microtus agrestis* persisted in suboptimal wet habitats where their main predator, the least weasel *Mustela nivalis nivalis* had a low hunting efficiency, so increasing the sub-optimal
habitat of field voles was predicted to increase their survival. A similar approach could be effective for eastern barred bandicoots by increasing the amount of predator refuge. Increasing habitat complexity could also be advantageous to eastern barred bandicoots if it reduced the hunting efficiency of red foxes. A reduction in European rabbits, the primary prey of the red fox, could theoretically allow bandicoots to be consumed according to a Type III functional response, in which few if any prey are taken at low densities (Sinclair et al. 1998). However, it is doubtful that the abundance of European rabbits would fall below that of bandicoots, so rabbit control could accelerate extinction of bandicoots. European rabbits are currently controlled at some reintroduction sites, and it would be advantageous to investigate the relationships between red foxes, European rabbits and eastern barred bandicoots.

The success of captive-bred versus wild-bred eastern barred bandicoots is another area for further research. Captive-bred bandicoots may be more susceptible to predation due to a lack of predator awareness. Short et al. (2002), reported that wild-bred burrowing bettongs appeared to survive longer than burrowing bettongs from red fox-free Dorre Island, Western Australia, or from the captive colony. If this is the case with captive-bred bandicoots, then individuals should be released into areas where low red fox numbers can be guaranteed. Alternatively, bandicoots could undergo predator training before release. This has been trialled in other marsupials (Griffin and Evans 2003a, Griffin and Evans 2003b, McLean et al. 1996, McLean et al. 1994) and birds (McLean et al. 1999, Maloney and McLean 1995), with the aim of linking fear or fright in the prey species with the image of the predator (McLean et al. 1994). There has been some success with this approach (e.g. McLean et al. 1999, McLean et al. 1996) and training is quick with little expenditure (McLean et al. 1996).

The number of animals in each release and the frequency of releases could have a significant effect on the success of a population. Boutin (1995) suggested that releases should be designed to increase abundance rapidly so that the population increases beyond regulation by predators. This prediction is based on predator prey theory and has been used by Sinclair et al. (1998) and in numerous models (e.g. McCallum et al. 1995), but field investigations are lacking. The number of eastern barred bandicoots in each release and frequency of releases throughout the bandicoot reintroduction program has been inconsistent. Thirty bandicoots were in the first release at Cobra Killuc (Table 2.1), and a further four releases totalling 73 bandicoots occurred over 26 months. This site was
unsuccessful and only one wild-bred bandicoot was captured there. Although other factors were involved in the demise of bandicoots at this site, their effect may have been less severe if all 103 bandicoots were released over a shorter time period. In contrast, a total of 85 bandicoots were released at Mooramong over 18 months (Table 2.1). This population quickly became established and is now relatively stable. On the other hand, 55 bandicoots were released at Lanark over 24 months. This population also established quickly, but later declined, presumably due to a reduction in predator control and ongoing dry conditions.

How animals are released into an area can also affect their survival. Two broad types of reintroductions exist: soft release (animals are held in protected enclosures at the release site and provided with food, water and shelter) and hard release (animals receive no post release support). Success has been achieved in a variety of animals for both soft release (e.g. Eastridge and Clark 2001) and hard release (e.g. Halley and Rosell 2002). Both types of release have occurred throughout the eastern barred bandicoot reintroduction program. Soft releases have involved acclimatisation pens, supplementary feeding and artificial hard shelters scattered throughout reserves. However, a study by Ferguson (2006) at Mt Rothwell, showed that acclimatisation did not increase survival of reintroduced captive-bred bandicoots and was deemed unnecessary when releasing bandicoots into predator-protected areas. Therefore, the extra time and costs involved with soft-release cannot be justified for eastern barred bandicoots.

Eastern barred bandicoots have been released into three sites surrounded by a predator-barrier fence. Mt Rothwell has had no red fox incursions since their initial removal. This could be attributed to the fence design, nightly fence inspections, ongoing fence maintenance and allowing public access only when accompanied by a ranger. Both Woodlands Historic Park and Hamilton Community Parklands have had predator control problems related to fence design or maintenance. Allowing public access has also caused problems. Gates are often left open and the fences have previously been vandalised. Red foxes and cats can still penetrate the Woodlands Historic Park perimeter by pushing past the electric hot wires at the top of the fence (Alan Robley pers. comm.). The fence design at Woodlands Historic Park is in need of improvement. Upgrades are continually being made and a skirt is currently being added. A ‘floppy top’ is also needed, but a lack of resources is currently preventing this. For similar reasons the Hamilton Community Parklands fence was upgraded in July 2005 at a cost of $30,000.
Chapter 2: Reintroductions

Predator-barrier fences are not necessarily required for successful bandicoot reintroductions. Mooramong has a healthy bandicoot population, possibly due to their predator control program being adequate for that site, and eastern barred bandicoots at the Grange Burn have persisted in the presence of presumably high predator densities and no specific management. Although this area is no longer monitored, sightings are reported occasionally and a road killed bandicoot was found in December 2005 (Richard Hill pers. comm.).

Conclusion

After 16 years of reintroductions and continuous management, the Victorian eastern barred bandicoot population remains small. An estimate of the current population size is not available, but it could be as few as 100 individuals. It is not clear which processes are influencing populations the most. A combination of drought and inadequate red fox control are the most likely limiting factors. It is possible that populations have declined due to a series of below average rainfall years, and that any increase in numbers has been inhibited by predation. Alternatively, it could be inadequate red fox control that initiated the decline at some sites, exacerbating the effects of drought on these small populations. As a reduction in red fox control intensity occurred concurrently with extended dry periods, our ability to draw robust conclusions about the processes involved is confounded.

Unless eastern barred bandicoot populations can escape limitation by predation it is unlikely that predator control can ever be withdrawn at reintroduction sites. Finding a cost-effective way to exclude or maintain low levels of introduced predators is essential. Fences play an important part in reintroductions, but the high cost involved in maintaining them results in less time and funds available for other important management issues, such as habitat management and predator control surrounding the fence. If predator-barrier fences are initially well designed and constructed, continual upgrades should not be necessary, reducing the overall cost of the fence. Involving the local community with predator control would also benefit bandicoot reintroductions. Community support and involvement may be politically important to convince stake-holders, including the government, to invest in the recovery of eastern barred bandicoots.

Investigations are currently underway to assess the relative importance of food availability and vegetative structure on eastern barred bandicoot population performance at Woodlands Historic Park, Hamilton Community Parklands and Mooramong. One of the
outcomes of this work will highlight if it is necessary to actively intervene at a site during certain abnormal conditions, such as below average rainfall. The eastern barred bandicoot Recovery Team is continuously reviewing site performance and has recently standardised monitoring across all sites. Annual trapping across the entire reserve will occur in spring to provide estimates of total population size when breeding has typically ceased (Hill 2006). Information on the proportion of each reintroduction site occupied by bandicoots will be collected quarterly by recording bandicoot digs within a 5 m radius at each permanent trap site (Hill 2006). This is intended to provide the most cost-effective data on shorter-term trends in population size.

Eastern barred bandicoot recovery is expensive and requires a continuous input of resources, without recognition and acceptance of this by the Government, the Victorian eastern barred bandicoot population may be restricted to a handful of small reintroduction sites indefinitely.
Chapter 3

An assessment of three techniques for monitoring a rare, cryptic grassland marsupial

A typical eastern barred bandicoot foraging dig at Hamilton Community Parklands, July 2007
Chapter 3

An assessment of three techniques for monitoring a rare, cryptic grassland marsupial

Abstract

Monitoring threatened species is often problematic due to their low population densities and resultant low detection probability. This study evaluated three low-cost monitoring techniques for the endangered eastern barred bandicoot *Perameles gunnii* at two reserves, Woodlands Historic Park and Hamilton Community Parklands, in Victoria, Australia. The former contained a low density population, whereas the latter had a relatively high density population of eastern barred bandicoots. I monitored both reserves by trapping using cage traps at plots on a regular grid, spotlighting in the same area as the trapping grid, and counting foraging digs within a 5-m radius of the centre of each plot. Trapping detected eastern barred bandicoots only in the high-density population. Spotlighting detected few eastern barred bandicoots at either site, probably because their small size and cryptic colouration obscured them when they were in small patches of vegetation. Digs were easily identified and counted at both reserves, and provided an index of population change at Hamilton Community Parklands. Dig density had a seasonal interaction with soil compaction, particularly in winter, but not with capture rate. Therefore it is recommended that dig counts occur in the winter months only. Using these three techniques together increased the probability of bandicoot detection and identified the growth trajectories of the populations.
**Introduction**

Population monitoring is essential for detecting changes in abundance or site occupancy, and identifying and managing threatened species (Clarke *et al.* 2003, Joseph *et al.* 2006). Any management actions undertaken must be monitored, so that the response of the population to the management action can be gauged; this requires the development of reliable monitoring methods (Clarke *et al.* 2003, Yoccoz *et al.* 2001).

Many different methods can be used to estimate population density, such as live trapping (e.g. Cunningham *et al.* 2005, Short and Turner 2000), camera traps (e.g. Karanth and Nichols 1998, Rowcliffe *et al.* 2008), hair traps (e.g. Mills *et al.* 2002, Catling *et al.* 1997) and distance sampling (e.g. Housman *et al.* 2005, Buckland *et al.* 2000), or to provide density indices such as track counts (e.g. Becker *et al.* 1998, Gusset and Burgener 2005) and faecal pellet counts (Shimano *et al.* 2006). Live trapping is particularly useful as it can generate population density estimates (Cunningham *et al.* 2005) using minimum number known to be alive or mark-recapture, as well as providing data on life history attributes. Although less informative, indices can provide information on presence, habitat-use and population density (e.g. Shimano *et al.* 2006). The same monitoring methods are applicable to the study of both common and rare species, but problems are inherent in monitoring low-density populations due to low detection probability. In the case of rare species, it is often necessary to combine techniques to increase the probability of detection (McDonald 2004). As recovery efforts are usually constrained by funding, developing a cost-effective monitoring regime is imperative.

Once widespread across the Basalt Plains of Victoria, the eastern barred bandicoot *Perameles gunnii* now survives in only a few managed, reintroduced populations on mainland Australia (Winnard and Coulson 2008). This species is a small (<1 kg), nocturnal marsupial, pale brown in colour, with three to four whitish bars across the rump; a cryptic pelage in its grassy woodland habitat. This species rests in a grass nest during the day and forages on invertebrates at night (Backhouse *et al.* 1994), creating characteristic conical-shaped digs up to a depth of 15 cm (Triggs 2001) but usually 4 - 6 cm deep (Dufty 1991b).

Eastern barred bandicoots were reintroduced into Woodlands Historic Park in 1989 (Winnard and Coulson 2008). The population was monitored by live-trapping for two consecutive nights three times a year. Once per year traps were set on a 100 x 200-m
permanent grid that covered the entire reserve, whilst twice yearly trapping concentrated on the area thought to have the highest densities of bandicoots (Katrina Lovett pers. comm.). In 1995 the population started to decline (Winnard and Coulson 2008). Since 1998, trapping has yielded very few captures and the last bandicoot was caught in April 2005 (Winnard and Coulson 2008). Monitoring by trapping continued unsuccessfully until 2007, although during this period characteristic foraging digs were observed and occasional bandicoots were seen during spotlight searches.

Hamilton Community Parklands was the second reintroduction site, established in 1989, but by 2004 this population was considered extinct (Winnard and Coulson 2008). After an upgrade of the predator-barrier fence, 24 bandicoots were released into the Hamilton Community Parklands in June 2007, and a further six were released in November 2007.

This study evaluates the effectiveness of three different techniques for monitoring a declining population at Woodlands Historic Park and an increasing population at Hamilton Community Parklands: trapping, spotlighting and foraging dig counts. These techniques were chosen as they are low cost and readily available to reserve managers. The usefulness of the data obtained was also assessed: known to be alive estimates from trapping, an index of population density from spotlighting, and another density index from foraging dig counts.

**Methods**

**Woodlands Historic Park**

Woodlands Historic Park is situated 22 km from Melbourne, Australia, and is a 300-ha grassy woodland reserve surrounded by a ‘leaky’ predator-barrier fence. The dominant grasses are kangaroo grass *Themeda triandra*, wallaby grass *Danthonia* spp., tussock grass *Poa labillardieri* and spear grass *Stipa* spp. The woodlands have a grassy ground layer and are dominated by river red gum *Eucalyptus camaldulensis*, but yellow box *Eucalyptus melliodora* is also common.

In winter 2006 I conducted bandicoot monitoring at 50 random plots (Fig 3.1) selected from the intersections of a 100 x 100-m grid based on the original 100 x 200-m permanent trapping grid. I baited standard 50 x 18 x 20-cm (length x width x height) cage traps with a
mixture of rolled oats, peanut butter and honey, placed a bundle of sea grass inside for insulation and protected them from the weather with plastic sleeves. I set one trap at each random plot for three consecutive nights, ceased trapping for four nights then resumed for a further three nights. I set traps in the late afternoon and checked them at dawn the next morning, recording all captures of target and non-target species.

When I captured a bandicoot I placed it into a fabric bag and identified it by scanning for a Trovan PIT (passive integrated transponder) tag, or inserted one if it was untagged. I then weighed it in the bag using a Pesola spring balance and examined the bandicoot for any parasites and injuries before turning it onto its back to sex and check female’s pouches. I counted any pouch young and measured their head length; if the pouch was empty I counted the number of lactating teats, if any. I then released the bandicoot from the bag and weighed the empty bag to subtract from the initial weight. These data took approximately 5 min to collect.

Fig 3.1: Woodlands Historic Park, showing the 50 random plots and the location of the 99 plot grid used to trap, spotlight and count foraging digs of eastern barred bandicoots from winter 2006 to summer 2007. Aerial photograph from Google Earth.

In spring 2006, I counted digs within a 3-m radius of each grid intersection on the 100 x 200-m permanent trapping grid. In the area of highest dig density I marked out a more
intensive 50 x 50-m grid containing 99 grid intersections, which formed the centre of each plot (Fig 3.1). I used this grid in spring 2006 and summer 2007. On the 100 x 200-m permanent trapping grid and the 50 x 50-m grid I counted digs within a 3-m and 5-m radius, respectively, by completing rotations around the centre of each plot, the first rotation counting digs between 0 and 1 m, the second between 1 and 2 m, and so on. I assumed that digs were recently made, as small disturbances to the soil readily changed their shape. Dig counts took an average of 10 min per plot. I calculated the number of digs per ha by dividing the number of digs in each plot by the area of the circular plot, then multiplied this figure by 10000 (e.g. 10 digs in a plot with a 5-m radius \((78.54 \text{ m}^2) = 0.127 \times 10000 = 1270 \) digs/ha). I then recorded five soil compaction (kg/cm²) readings, one at the centre of the plot, and one at each of the four cardinal directions, 10 m from the plot centre, using a pocket penetrometer (Geotest Instrument Corp), taking the mean of the five readings. At 1 m from each plot centre and at a different cardinal direction each season, I determined soil moisture by taking a 10-cm deep soil core, as no digs in this study exceeded a depth of 10 cm. I removed any organic matter and calculated the percentage soil moisture by weighing the sample pre- and post-drying in an oven at 105°C for 24 h (Australian Standard AS 1289.2.1-2005).

I conducted spotlight surveys for two nights during winter 2006 and summer 2007. I surveyed only the areas containing the highest density of digs in winter 2006, and surveyed the entire 50 x 50-m grid in summer 2007 by walking transects north-south along the grid with a 12-V, 55-W quartz halogen spotlight. I began the surveys 1 h after dark and continued for 2 h. I recorded the number and location of all eastern barred bandicoot observations.

Hamilton Community Parklands

Hamilton Community Parklands is situated on the outskirts of the city of Hamilton, 280 km west of Melbourne. This 100-ha grassy woodland reserve is surrounded by a predator-barrier fence that was upgraded in 2005. The dominant grasses are kangaroo grass, spear grass and Phalaris spp., and the most common trees are black wattle Acacia mearnsii, swamp gum Eucalyptus ovata and river red gum.

Monitoring commenced one week after the initial release in June 2007, on a 100 x 150-m grid covering the entire reserve (Fig 3.2). I used the same trapping protocol as at Woodlands Historic Park, except that I placed two traps (a total of 120 traps per night) at
each grid intersection, which formed the centre of each plot, in an attempt to compensate for non-target captures, and I used a different trapping frequency. In winter 2007, I set traps for two consecutive nights, and then for a further three consecutive nights two weeks later, so the health status of the released eastern barred bandicoots could be closely monitored. From spring 2007 to autumn 2008, I set traps for three consecutive nights unless prevented by extreme heat or stormy weather.

I conducted spotlight surveys on one night each season, with the exception of autumn 2008 and summer 2009. I drove along all the vehicle tracks (approximately 7 km), in a 4WD utility vehicle, at roughly 10 km h⁻¹ with a 12-V, 55-W spotlight held out the window. I conducted systematic dig searches at each plot in all seasons using the same methods as at Woodlands Historic Park, and measured soil compaction and soil moisture at each plot using the same techniques.

Fig 3.2: Hamilton Community Parklands, showing the location of the 60 plots used to trap, spotlight and count foraging digs of eastern barred bandicoots from winter 2007 to autumn 2009. Aerial photograph from Google Earth.
**Data Analyses**

The minimum number of bandicoots known to be alive was used as an index of density. This was calculated as per Slade and Blair (2000): a count of bandicoots trapped at the session of interest plus individuals trapped before and after but not during the session of interest. Despite the limitations of KTBA (Jolly and Dickson 1983, Nichols and Pollock 1983, Pollock et al. 1990) this method was chosen as some trap sessions were conducted for two consecutive nights, so were unsuitable for mark-recapture estimates. KTBA is useful as a lower limit of population size (Jolly and Dickson 1983) therefore this method was favoured over other techniques that do not generate a value for abundance, such as an index using captures per trap night.

The relationship between soil moisture, soil compaction, capture rate (the number of bandicoots captured at each location divided by the number of trap nights in that monitoring session: Hamilton only) and bandicoot foraging digs was examined using linear mixed models (LMM) fitted in the statistical package GenStat 12. Due to the differences in area sampled in each season at Woodlands (50 random plots in winter 2006 and a 99 plot grid in spring and summer 2007), the data were not combined into one model. Therefore I split the data and ran a model on the winter 2006 data and a second model on the spring and summer 2007 data. All data collected at Hamilton Community Parklands was placed into the same model. For all three models, any plots that contained zero digs were removed from the analysis to prevent biased detection error. This is an inherent problem in habitat selection studies, particularly when cryptic species are involved (Rodhouse et al. 2010); non-detection of the species is not equivalent to absence because a species can go undetected when present (MacKenzie et al. 2002, Tyre et al. 2003). In a low density population, a patch may be suitable for foraging but there are not enough bandicoots in the reserve to fill that patch or, in the case of a newly reintroduced population, bandicoots may not have had time to reach and occupy that patch.

In all three models the foraging dig data were log_e transformed to better satisfy the assumptions of constant variance and normality of the errors. The Woodlands foraging dig data were still skewed to the right (i.e. not normally distributed) after a log_e transformation due to the small dig counts in most plots, especially the large number of 1s. For example in winter 2006 only 13 out of 50 plots contained digs; 11 of these plots contained four or less digs and the remaining two contained 22 and 66 digs. A similar pattern was observed in the
spring 2006 and summer 2007 data. Prior to running the models I calculated any correlations between explanatory variables; soil compaction was reasonably correlated with soil moisture in the Woodlands spring 2006 - summer 2007 data ($r = +0.52$) and the Hamilton data ($r = +0.56$). In all three models ‘$\log_e$ digs’ was the response variable and ‘plot’ was a random factor. I then added individual variables to the model one by one to investigate if there was a significant ($P < 0.05$) interaction with the seasonal effect (except for the Woodlands winter 2006 data as there was only one season contained within this dataset). Only soil compaction at Hamilton showed a significant interaction. I then added individual variables to this significant interaction. Any variable that did not make a significant ($P < 0.05$) contribution to the model was removed. The Woodlands winter 2006 model contained no significant explanatory variables, whilst the spring 2006 to summer 2007 model contained only soil moisture and the final Hamilton model contained the soil compaction by season interaction and capture rate.

**Results**

*Woodlands Historic Park*

No bandicoots were trapped, only one individual was observed by spotlight and 16% of plots contained digs in winter 2006 (Table 3.1). After the sampling design was modified, the proportion of plots containing digs doubled in spring 2006, and then declined to previous levels in summer 2007. This change in sampling design prevents any direct comparisons between winter 2006 and spring 2006 and summer 2007. The mean density of digs was highest in spring 2006 and lowest in summer 2007, coinciding with an increase in soil compaction and a decrease in soil moisture.
Table 3.1: Eastern barred bandicoot monitoring data for Woodlands Historic Park from winter 2006 to summer 2007. Means are ± standard error.

Data from winter 2006 are from 50 random plots and spring 2006 and summer 2007 data are from a 99-plot grid. Trapping and spotlighting was not conducted in spring 2006.

<table>
<thead>
<tr>
<th></th>
<th>No. trap nights</th>
<th>Known to be alive</th>
<th>No. spotlighted</th>
<th>% plots with digs</th>
<th>Mean digs/ha</th>
<th>Mean soil moisture (%)</th>
<th>Mean soil compaction (kg/cm²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter 06</td>
<td>300</td>
<td>0</td>
<td>1</td>
<td>16</td>
<td>290 ± 176</td>
<td>13.6 ± 0.7</td>
<td>1.0 ± 0.13</td>
</tr>
<tr>
<td>Spring 06</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td>34</td>
<td>329 ± 94</td>
<td>4.8 ± 0.1</td>
<td>1.7 ± 0.84</td>
</tr>
<tr>
<td>Summer 07</td>
<td>594</td>
<td>0</td>
<td>0</td>
<td>16</td>
<td>46 ± 15</td>
<td>0.3 ± 0.1</td>
<td>2.4 ± 0.10</td>
</tr>
</tbody>
</table>

In winter 2006, soil compaction and soil moisture were not significantly related to the number of bandicoot foraging digs ($P = 0.86$ and 0.50 respectively). In spring 2006 and summer 2007, soil moisture had a significant ($P <0.001$) relationship with the number of bandicoot foraging digs but there was no interaction with season. The output of the model fits into the general equation:

$$\log_e \text{digs} = a + b \times \text{soil moisture}$$

Where $a$ is the intercept (0.95) and $b$ is the soil moisture coefficient (0.20). The standard error of this coefficient is 0.05.

To allow the model output to be more easily interpreted, the coefficients for soil moisture were back transformed to the original scale and converted into a percentage. As soil moisture increased by 1% the number of digs is estimated to increase by a factor of $e^{0.20} = 1.22$ or 22%, regardless of season.

Hamilton Community Parklands

The number of bandicoots known to be alive was relatively stable throughout the study, ranging from 22 to 38, with the exception of spring 2008, when 53 were known to be alive (Table 3.2). Spotlighting yielded a maximum of nine sightings in spring 2008. The number of plots containing digs increased until winter 2008, with the exception of autumn 2008, then fluctuated between 92% and 97% for the remainder of the study. Mean density of digs
appeared to be related to soil moisture and compaction, with declines in dig numbers coinciding with a decline in soil moisture and an increase in soil compaction, although every season did not strictly adhere to this pattern.

In the final model all $P$ values were <0.01. The interaction between soil compaction and season was highly significant ($P = 0.005$). Therefore different equations were constructed for each season. The interaction of season with soil compaction was greatest in winter 2008, with a decrease in soil compaction resulting in more digs. In the other seasons the same pattern was observed, but the coefficient (slope) for soil compaction (Table 3.3) was very close to zero, indicating that soil compaction was having little effect on digs in those seasons. There was no seasonal interaction for capture rate ($P <0.001$ for all seasons combined).

Table 3.2: Eastern barred bandicoot monitoring data for Hamilton Community Parklands from winter 2007 to autumn 2009. Means are ± standard error.

Spotlighting was not conducted in autumn 2008 or summer 2009.

<table>
<thead>
<tr>
<th></th>
<th>No. trap nights</th>
<th>Known to be alive</th>
<th>No. % plots with digs</th>
<th>Mean digs/ha</th>
<th>Mean soil moisture (%)</th>
<th>Mean soil compaction (kg/cm$^2$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter 07</td>
<td>600</td>
<td>22</td>
<td>1</td>
<td>1867 ± 192</td>
<td>30.4 ± 0.7</td>
<td>0.5 ± 0.04</td>
</tr>
<tr>
<td>Spring 07</td>
<td>358</td>
<td>23</td>
<td>2</td>
<td>456 ± 141</td>
<td>20.2 ± 0.7</td>
<td>2.2 ± 0.11</td>
</tr>
<tr>
<td>Summer 08</td>
<td>360</td>
<td>32</td>
<td>1</td>
<td>318 ± 51</td>
<td>10.4 ± 0.5</td>
<td>1.9 ± 0.11</td>
</tr>
<tr>
<td>Autumn 08</td>
<td>240</td>
<td>32</td>
<td>n/a</td>
<td>259 ± 46</td>
<td>8.1 ± 0.7</td>
<td>2.9 ± 0.12</td>
</tr>
<tr>
<td>Winter 08</td>
<td>450</td>
<td>33</td>
<td>4</td>
<td>3045 ± 284</td>
<td>23.4 ± 0.6</td>
<td>1.0 ± 0.05</td>
</tr>
<tr>
<td>Spring 08</td>
<td>360</td>
<td>53</td>
<td>9</td>
<td>1651 ± 208</td>
<td>9.1 ± 0.5</td>
<td>2.5 ± 0.14</td>
</tr>
<tr>
<td>Summer 09</td>
<td>240</td>
<td>35</td>
<td>n/a</td>
<td>1063 ± 144</td>
<td>10.9 ± 0.4</td>
<td>3.1 ± 0.13</td>
</tr>
<tr>
<td>Autumn 09</td>
<td>360</td>
<td>38</td>
<td>7</td>
<td>2012 ± 231</td>
<td>15.7 ± 0.5</td>
<td>2.4 ± 0.11</td>
</tr>
</tbody>
</table>
Table 3.3: Output of the linear mixed model showing the seasonal estimated coefficients, standard errors (SE) and back transformed coefficients, converted to a percentage, for all significant variables collected from Hamilton Community Parklands between winter 2007 and autumn 2009.

Values in the table fit into the general equation:

\[
\text{Log}_e \text{digs} = a + b \times \text{soil compaction} + c \times \text{capture rate}
\]

<table>
<thead>
<tr>
<th></th>
<th>Intercept ((a))</th>
<th>Coefficient ((b))</th>
<th>SE</th>
<th>%</th>
<th>Coefficient ((c))</th>
<th>SE</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter 2007</td>
<td>2.28</td>
<td>0.20</td>
<td>0.56</td>
<td>22</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Spring 2007</td>
<td>1.25</td>
<td>-0.52</td>
<td>0.56</td>
<td>-41</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Summer 2008</td>
<td>0.78</td>
<td>-0.15</td>
<td>0.56</td>
<td>-14</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Autumn 2008</td>
<td>0.94</td>
<td>-0.35</td>
<td>0.56</td>
<td>-29</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Winter 2008</td>
<td>1.79</td>
<td>-1.06</td>
<td>0.56</td>
<td>-65</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Spring 2008</td>
<td>2.18</td>
<td>-0.16</td>
<td>0.56</td>
<td>-15</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Summer 2009</td>
<td>1.65</td>
<td>0.06</td>
<td>0.56</td>
<td>6</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
<tr>
<td>Autumn 2009</td>
<td>2.48</td>
<td>-0.07</td>
<td>0.56</td>
<td>-7</td>
<td>0.51</td>
<td>0.14</td>
<td>67</td>
</tr>
</tbody>
</table>

The coefficients for soil compaction and capture rate were back transformed to the original scale and converted into a percentage (Table 3.3). For every 1 kg/cm² increase in soil compaction in winter 2008, the number of digs was predicted to change by a factor of \(e^{-1.06} = 0.35\) or a 65% decline. For most of the other seasons, the number of digs was predicted to decline, but at a slower rate. As capture rate increased by 0.1 unit, the number of digs was estimated to increase by a factor of \(e^{0.51 \times 0.1} = 1.05\) or 5%, regardless of season.

**Discussion**

**Trapping**

Trapping is a common technique used to monitor small mammals (Cunningham *et al.* 2005) and has been successfully used for eastern barred bandicoots (Dufty 1991b), as well as other bandicoot species (Mills *et al.* 2002). Trapping was the only monitoring technique used at Woodlands Historic Park prior to this study. This technique resulted in numerous bandicoot captures when the population size was large, but once the population declined,
very few or no captures occurred. In this study no bandicoots were trapped at Woodlands Historic Park despite bait type and trap placement being similar to previous, successful monitoring events, and an increase in trapping intensity (1488 trap nights in three seasons). This is indicative of a very low population density. At Hamilton Community Parklands, a reserve with an increasing population, capture rate was much higher than at Woodlands Historic Park in each survey, despite traps being spaced further apart and the trapping intensity being reduced.

**Spotlighting**

Spotlighting is widely used for assessing presence and density of nocturnal species (Scott et al. 2005). A range of nocturnal mammals, such as the Eurasian badger *Meles meles* and red fox *Vulpes vulpes* are surveyed by distance sampling using spotlights, which offers a simple, efficient technique (Hounsome et al. 2005). However, studies have found spotlighting to have low precision and species detection bias (Scott et al. 2005), particularly at low population densities in woodland areas (Hounsome et al. 2005, Mahon et al. 1998). In this study, spotlighting had the lowest detection rate of the three techniques used. Only one bandicoot was ever observed during spotlight searches at Woodlands Historic Park. Sightings occurred on every occasion at Hamilton Community Parklands and a maximum of nine bandicoots was observed on any night. Although the methods used at each site varied slightly, being conducted on foot at Woodlands Historic Park and from a vehicle at Hamilton Community Parklands, the higher detection rate at Hamilton Community Parklands was clearly attributable to the larger population size. However, there were still only a few sightings, and these were generally opportunistic, as occurred when bandicoots were disturbed by the vehicle. Due to the lack of sensitivity, this technique is unsuitable for monitoring eastern barred bandicoots. However, it could be useful as a low-cost technique to detect presence, habitat occupancy and substantial population changes, but only if combined with the regular and continuing red fox spotlighting that occurs at these sites.

**Dig counts**

Indirect counts, such as faecal pellet counts, are regularly used as indices of habitat use and population density (Shimano et al. 2006). Other indirect indices such as burrow counts (Smallwood and Erickson 1995) have been developed to monitor population change. Data on habitat use, such as foraging patch preference, can also be obtained from these counts. Mallick et al. (1997b) developed an abundance estimate of the Tasmanian form of the
eastern barred bandicoot, using foraging digs. In this study, digs were correlated with capture rate but there was no seasonal interaction (Table 3.3). In addition to this, the number of digs increased with less compact soils; this effect was seasonal and strongest in winter 2008. Therefore, digs can be used as an index of population change, but not an estimate of abundance, and I recommend conducting dig counts in the winter months when they are more prolific.

Both reserves showed a decline in the number of digs from the coolest to the warmest months. This could indicate a population decline at Woodlands Historic Park but not at Hamilton Community Parklands, as the number of eastern barred bandicoots known to be alive was relatively stable across all seasons (Table 3.2). Changes in soil compaction had the greatest effect on the number of foraging digs in winter 2008 and very little effect in the other seasons (Table 3.3). This could be due to a reduction of digging by bandicoots due to the downward movement of macro-invertebrates in the soil profile during dry, hot weather (Staley et al. 2007). Bandicoots would then be forced to switch to foraging from the soil surface, which leaves no visible sign. An increase in the abundance of above-ground invertebrate prey in the warmer months (Gibson 2001) may also reduce the need to dig.

Conclusion

This study evaluated low-cost monitoring techniques that are readily available to managers of eastern barred bandicoot reserves. A combination of these techniques could be used as management triggers. A rapid decline in trap success could trigger investigations into the decline, whilst a rise in spotlight sightings could provide the first indication of a recovering population. The foraging patch preference of eastern barred bandicoots can also be easily identified using dig counts, particularly in winter months. Other techniques, such as hair-traps (Fonturbel 2010) and camera-traps (Rowcliffe et al. 2008) could be considered for monitoring this rare, cryptic species at low densities. However, when population density is as low as that observed at Woodlands Historic Park, a more appropriate management response would be to devote resources to tackling the cause of the decline, rather than seeking alternative monitoring techniques. Although these survey techniques were developed specifically for the eastern barred bandicoot, this management approach could be applied to monitoring other rare or elusive species. Using complementary techniques with multiple triggers increases detection probability and clarifies the growth trajectory of the population under study, resulting in more appropriate management actions.
Chapter 4

Living with the enemy: dynamics of a declining population of the critically endangered eastern barred bandicoot *Perameles gunnii*

*A typical red fox in Victoria (photograph by Jaco Coetsee)*
Chapter 4

Living with the enemy: dynamics of a declining population of the critically endangered eastern barred bandicoot *Perameles gunnii*

Abstract

Understanding how vital rates change and affect population growth is particularly important for conservation management, so that factors causing declines can be minimised, if not removed. This study analyses a 17-year data set obtained from monitoring a reintroduced population of eastern barred bandicoots *Perameles gunnii* at Mooramong, Victoria, Australia. This population has persisted longer than any other reintroduction site despite red foxes *Vulpes vulpes*, the main predators of eastern barred bandicoots, not being eradicated from the reserve. This population showed some evidence of an Allee effect. Air temperature was found to negatively affect the proportion of females breeding and their litter size, with a 2-month lag, possibly due to the extended feeding times permitted in the cooler months and easily accessible soil invertebrates. Mortality is considered to be the main component causing the Allee effect and regulating this population. The main cause of this mortality is likely to be predation by introduced red foxes, which was exacerbated during a prolonged drought in the mid 1990s.
Introduction

Conservation management of any population critically depends on the factors affecting vital rates and driving population growth (Sinclair and Krebs 2002). These factors can be either density dependent or density independent. Density dependent effects are generally considered to be the outcome of resource availability, competition, or predation (Klinger 2007), whilst factors such as climate and disturbance are density independent (Previtali et al. 2009).

Density dependent effects can be negative or positive. For density to have a negative effect on population growth, a vital rate such as fecundity must be negatively affected by density (Reed and Slade 2008). This can occur through a behavioural or physiological reaction to rising density, which causes a decline in resource availability (Caughley and Krebs 1983). Two possible responses to high density are delayed reproduction and reduced litter size (Reed and Slade 2008). Density dependent mortality is usually attributed to increased competition for food at high densities, which can decrease body mass and negatively affect adult and juvenile survival (Bonenfant et al. 2009). Predation and parasitism can act as additional stressors contributing to mortality, although the actual cause of mortality can be hard to decipher (Bonenfant et al. 2009). At low densities, Allee effects can have a positive feedback effect on population growth, increasing extinction risks by reducing vital rates (Deredec and Courchamp 2007). From a conservation perspective, mate-finding Allee effects are of most concern, as they have the potential to lead to a catastrophic population collapse and prevent or slow recovery from low densities (Gascoigne et al. 2009).

Density independent climatic conditions, such as temperature and precipitation, have been shown to affect densities of degu Octodon degus and Darwin’s leaf-eared mouse Phyllotis darwini (Previtali et al. 2009) and the little bustard Tetrax tetrax (Delgado et al. 2009), and nest success, clutch size and the number of chicks fledged in the woodlark Lullula arborea (Wright et al. 2009). Rainfall in particular has been found to have strong density independent influences on the vital rates of survival, condition, fecundity and overall abundance of numerous species, by altering the availability of food, such as insect and seed abundance (Sinclair and Krebs 2002, Chase et al. 2005, de Lima and Roper 2009, Sale et al. 2008).

Small mammal populations are rarely limited by food supply, but are more frequently effected by predation pressure, coupled with social interactions (Sinclair and Krebs 2002). If
predation pressure is applied by an introduced predator, it can have drastic impacts on native prey populations, which often lack the appropriate behavioural traits to avoid introduced predators (Fey et al. 2008). Indeed, in reintroductions where introduced predators have not been completely removed, the program has often failed (Sinclair et al. 1998). Krebs (2009) classified introduced predators as a form of habitat degradation, which is density independent, arguing that there would not be intrinsic population control when such predators were present.

The eastern barred bandicoot *Perameles gunnii* is critically endangered (Victorian DSE 2007) in Victoria, Australia, and survives only in reintroduced populations (Winnard and Coulson 2008). This small (<1 kg), nocturnal, solitary omnivore has been driven to near extinction by a >99% loss of native grasslands and grassy woodlands, in which this species formerly occurred, and by predation, in particular the introduced red fox *Vulpes vulpes* but also the feral cat *Felis catus* (Backhouse et al. 1994). The eastern barred bandicoot is highly fecund and can breed throughout the year, with females producing up to five litters per year with an average of two to three young in each litter (Backhouse et al. 1994). A life span of 2 - 3 years is typical (Backhouse et al. 1994), and average densities of 1.5 individuals ha⁻¹ have been reported (Minta et al. 1990). Since 1989, eight reintroduction sites have been established in Victoria with varying success in terms of the survival and reproduction of released bandicoots and reproduction of wild-bred bandicoots (Winnard and Coulson 2008).

In this study I analysed a 17-year data set of an eastern barred bandicoot population reintroduced into Mooramong, in Western Victoria. This population has persisted longer than any other reintroduced population despite the presence of red foxes, which have been suppressed but not eradicated from the site (Winnard and Coulson 2008). I aimed to determine the relative importance of density dependent and density independent mechanisms on the population rate of change.

**Methods**

**Study Site**

Mooramong is a working sheep farm, 170 km west of Melbourne, Australia. The area has an average yearly rainfall of 622 mm and temperatures range from a mean annual minimum of
7.0°C and a maximum of 17.3°C. Climate data were obtained from the Bureau of Meteorology (www.bom.gov.au); rainfall data from the Skipton Post Office (~10 km south-east of Mooramong) and air temperature data from Ballarat Aerodrome (~50 km north-east of Mooramong).

A total of 85 eastern barred bandicoots were released on the property into a 243-ha nature reserve, between 1992 and 1994 (Winnard and Coulson 2008). These bandicoots were sourced from captive breeding pens within the Nature Reserve. Regular red fox control occurred throughout the year by poisoning with 1080 (sodium monofluoroacetate) baits, den fumigation and shooting. Three areas within Mooramong were occupied by bandicoots over the course of this study: the Nature Reserve, Cottage Dam and Homestead Gardens (Fig 4.1). The Nature Reserve and adjacent Cottage Dam area is dominated by introduced pasture grasses such as canary grass *Phalaris aquatica* and Yorkshire fog *Holcus lanatus*, with a sparse open woodland of silver banksia *Banksia marginata*, drooping sheoak *Allocasuarina verticillata*, blackwood *Acacia melanoxylon* and sweet bursaria *Bursaria spinosa*. Both areas contain ephemeral wetlands, which from 1996 have been dry ~85% of the time. The Homestead Gardens are highly modified with mowed lawns bordered by beds of *Agapanthus orientalis* and stands of Monterey Pine *Pinus radiata*. Native predators of bandicoots at Mooramong include wedge-tailed eagles *Aquila audax* and tiger snakes *Notechis scutatus*.

![Fig 4.1: Mooramong, showing the location of the Nature Reserve, Cottage Dam and Homestead Gardens. Aerial photograph from Google Earth.](image)
Chapter 4: Dynamics of decline

I used data obtained from live-trapping eastern barred bandicoots between December 1992 and October 2009. Trapping was conducted by staff of the Department of Sustainability and Environment and its precursors, occurring every 31 ± 2.4 days (mean ± SE) from December 1992 to April 1994, after which the interval increased to 100 ± 5.1 days. In general, trap sessions occurred over two consecutive nights, and trap placement focused on maximising capture rate rather than spatial replication. This resulted in a gradual reallocation of effort from the Nature Reserve to the Cottage Dam, then the Homestead Gardens in December 1997, in response to an apparent redistribution of the population. In the Nature Reserve, traps were placed on random transects selected from a permanent 50 x 50-m grid. In the Cottage Dam and Homestead Gardens, trap placement was at unmarked haphazard locations; where location data exist, there appears to be some consistency in trap placement between monitoring sessions, but these locations were often labelled differently in each trap session. Traps were 50 x 18 x 20-cm (length x width x height) cage traps baited with peanut butter, honey and rolled oats, which were set in the late afternoon and checked early the next morning. Captured bandicoots were marked with unique ear tattoos until 1995, after which Trovan PIT (passive integrated transponder) tags were used. The sex and weight of adults and the number of lactating teats for each female was recorded in addition to the number and head lengths of pouch young.

Data Analysis

I omitted data collected before 1994 from all analyses, except disappearance rate, as there was ambiguity regarding the location of the bandicoot captures (i.e. if they were caught in the captive breeding pens or free ranging). There were 1,400 captures during the study, comprising 297 individual bandicoots (139 female and 158 male). Of these, 38% were caught only once.

I did not separate the data by age class as adult eastern barred bandicoots cannot be aged by morphological criteria. Therefore, I was unable to calculate survival rates and longevity. Instead I calculated the disappearance rate for males and females, measured as the total number of months over which an individual was trapped. Bandicoots that were caught on only one occasion were not included in this analysis. I then conducted a Kolmogorov-Smirnov two-sample test, in GenStat 12, on bandicoots captured more than once to compare the disappearance rate of males and females.
Due to the lack of randomisation and replication of monitoring surveys, any statistical estimate of density would result in unreliable parameter estimates and uninformative or misleading inferences (Williams et al. 2002). Therefore, I calculated the minimum number of bandicoots known to be alive (KTBA) at each monitoring session. This was calculated as per Slade and Blair (2000): a count of individual bandicoots trapped at the session of interest plus individuals trapped both before and after but not during the session of interest.

Reproductive data such as number and head length of pouch young and number of lactating teats was collected from females. I used only the presence and number of pouch young in analyses. Lactating females that had young in a nest were not included in analyses as the data entry for this variable was sporadic. For each trap session I calculated the proportion of adult females with pouch young and the mean litter size. I defined the minimum weight of an adult as the mean female breeding weight (795 g) and subtracted 2 standard deviations (235 g); three females with pouch young fell below this limit and were omitted from the analysis.

I calculated the exponential rate of increase ($r$) between trap sessions as $r = \log_e(N_t/N_i)$ (Caughley 1980). I then examined linear regression between rate of increase and KTBA for every trap session post 1994. I also used these data and linear regression to examine the relationship between KTBA and proportion of females with pouch young and mean litter size. I then performed standard multiple regression analyses in SPSS 17.0 to determine the effect of mean monthly precipitation and mean monthly temperature on rate of increase. I lagged the climatic data by 1, 2 and 3 months, as it was expected that resources such as food and cover would not respond immediately to a change in climatic conditions. Therefore a delayed response would also be observed in bandicoots.

I fitted two generalised linear mixed models (GLMMs) in GenStat 12 to determine the relationship of mean monthly precipitation and mean monthly temperature on (i) litter size, using a Poisson distribution and (ii) breeding using a binominal distribution. In both models I lagged the climatic data by 1, 2 and 3 months. Individual female bandicoots and trap session were treated as random factors in both models. In the litter size model, only females with pouch young where used, whilst in the breeding model, ‘breeding’ was a categorical response variable: breeding (i.e. with pouch young) or not breeding. Lactating females, with young in a nest, were not used in either model. I constructed the most
parsimonious model by deleting terms that did not make a significant ($P < 0.05$) contribution to the model. Using this criterion, precipitation with a 1, 2 and 3-month lag was deleted from both models. In the litter size model, temperature with a 1, 2 or 3-month lag was significant when each was placed into the model individually, but when each temperature variable was combined with another, only temperature with a 2-month time lag was significant. In the breeding model, temperature with a 1-month time lag was not significant, but temperature with a 2 and 3-month lag were, even when placed in the model together. However, these temperature variables were highly correlated, so temperature with a 3-month lag was removed from the model as temperature with a 2-month lag was more significant. Therefore, the final model for both litter size and breeding contained only temperature with a 2-month lag.

**Results**

The population fluctuated widely and was never in a continual state of growth: 54% of monitoring sessions revealed a population decline, 10% showed no change in growth and 36% showed an increase (Fig 4.2). The population reached its maximum in April 1994, when 50 bandicoots were known to be alive (Fig 4.3). This was followed by a sharp decline, which reached its lowest point in September 1995 (14 bandicoots KTBA) before the population increased, peaking at 38 KTBA in March 1998. Up to this time all bandicoots had been trapped in the Nature Reserve, or the Cottage Dam, but capture rates were low (<15 bandicoots caught per 100 trap nights) in both areas (Fig 4.4). Bandicoots appeared in traps set in the Homestead Gardens the first time they were placed there in December 1997, but this is not shown in Fig 4.4 as there was no record of how many traps were set during that trap session. A second decline was evident in June 1998, and in September 1999 only five bandicoots were known to be alive (Fig 4.3). Again the population recovered, but reached only a maximum of 22 bandicoots KTBA in September 2001 (Fig 4.3). The majority of these captures occurred in the Homestead Gardens, whilst the Nature Reserve population remained low and the last bandicoot was caught there in June 2004 (Fig 4.4). From 2001, the population gradually declined, and no bandicoots were caught in the last trap session in October 2009 (Fig 4.3).
Six bandicoots were detected moving between the Nature Reserve and Homestead Gardens (Fig 4.5). Four of these (3 male and 1 female) moved to the Homestead Gardens, either prior to or during the 1998/99 population crash, another two males moved after this crash (Fig 4.5). One of these males moved from the Nature Reserve to the Homestead Gardens whilst the other moved in the opposite direction. None of the six bandicoots was known to be alive before and after the crash; four individuals (3 male and 1 female) disappeared either prior to or during the crash, whilst the other two appeared after the population had begun to recover (Fig 4.5).

**Fig 4.2**: Population rate of increase of eastern barred bandicoots at Mooramong between consecutive trap sessions from January 1994 to March 2009.
Chapter 4: Dynamics of decline

Fig 4.3: Number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to March 2009.

Fig 4.4: Number of eastern barred bandicoots caught per 100 trap nights at Mooramong in the Nature Reserve, Cottage Dam and Homestead Gardens from February 1995 to October 2009. Data were unavailable for months marked with an asterix.
Of the 297 bandicoots captured at Mooramong, 50% were captured less than three times and only 0.7% were captured more than ten times. Females ($\bar{x} = 3.41, SD = 2.79$) were captured more often than males ($\bar{x} = 2.81, SD = 2.12; t = -2.06, df = 255.176, P < 0.05$). The last female captured at Mooramong was in February 2008 and the last male in March 2009. The low capture rate was coupled with a high disappearance rate (Fig 4.6). More than 50% of males ($n = 108$) and females ($n = 99$) disappeared after 10 and 15 months respectively ($\chi^2 = 9.74, df = 2, P = 0.008$). The longest a male and female bandicoot was known to live was 32 and 39 months respectively (Fig 4.6).

Since January 1994, 528 pouch young were recorded in 248 litters; 46 of these litters comprised of only one young, 135 had two, 56 had three and 11 had four. The percentage of females with pouch young fluctuated between 100% and 0%. No pouch young were recorded for any female during five trapping sessions, between the months of January and April corresponding with mid summer to early autumn. In general the number of pouch young fluctuated between 1.5 and 2.5 per female, with a mean of 1.9, and varied seasonally with means ± SE of 2.6 ± 0.1 ($n = 73$) pouch young per litter in spring, 1.8 ± 0.1 ($n = 52$) in summer, 1.8 ± 0.1 ($n = 40$) in autumn and 2.1 ± 0.1 ($n = 83$) in winter. The highest mean litter size was 3.4 in September 2002.
There was a weak, positive relationship between KTBA and rate of increase ($r^2 = +0.17$, $df = 58$, $P = 0.001$; Fig 4.7). There was also a weak significant relationship between KTBA and the proportion of females with pouch young ($r^2 = -0.10$, $df = 57$, $P = 0.016$; Fig 4.8) and mean litter size ($r^2 = -0.10$, $df = 51$, $P = 0.026$; Fig 4.9).

**Fig 4.7**: Linear regression of the population rate of increase on the number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to June 2008.
Fig 4.8: Linear regression of the proportion of females with pouch young on the number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to June 2008.

Fig 4.9: Linear regression of mean litter size on the number of eastern barred bandicoots known to be alive (KTBA) at Mooramong from January 1994 to June 2008.
Mean monthly precipitation with a 1, 2 and 3-month lag had virtually no effect on the population rate of increase \( (r^2 = +0.006, +0.010 \text{ and } +0.014 \text{ respectively}) \). Mean monthly temperature with a 1 and 2-month lag made a very weak but significant contribution to rate of increase \( (r^2 = +0.079, P = 0.034 \text{ and } r^2 = +0.082, P = 0.031 \text{ respectively}) \), whilst mean monthly temperature with a 3-month lag made very little contribution \( (r^2 = +0.032, P = 0.180) \). Combining rainfall and temperature for each monthly lag did not increase the amount of explained variance, nor did combining the three rainfall or temperature variables in one model.

In the final models (GLMMs), both litter size and breeding was negatively correlated with temperature with a 2-month lag. In the litter size model the coefficient of temperature was \( -0.034 \) (standard error \( \pm 0.0063 \)). Therefore, for each additional degree of temperature with a 2-month lag, \( \log_e \) pouch young was estimated to decrease by a factor of \( e^{-0.034} = 0.97 \) or 3.3%. In the breeding model a calculation of the odds ratio reflects the strength of the correlation: for each additional degree of temperature with a 2-month lag, the odds of breeding was predicted to decrease by a factor of \( e^{-0.241} = 0.79 \) (95% confidence interval: 0.72, 0.86) or 21%.

**Discussion**

Between 1992 and 1994, 85 bandicoots were released into the Mooramong Nature Reserve. Since these releases, this population has never been in a continual state of growth (Fig 4.2). In April 1994 the population reached its peak, with 50 bandicoots known to be alive (KTBA) but then declined immediately (Fig 4.3). The population later recovered before a second decline in 1998 (Fig 4.3). Again the population recovered slightly but the KTBA gradually declined (Fig 4.3), and in October 2009 no bandicoots were captured. This indicates that mortality, whether juvenile, adult or both, repeatedly exceeded fecundity.

A demographic Allee effect was evident, because as KTBA declined, so too did the population growth rate, which fell below zero at 24 individuals (Fig 4.7) (Stephens *et al.* 1999). However, this effect was not caused by the two measured fecundity components, the proportion of females with pouch young (Fig 4.8) and their mean litter size (Fig 4.9), because both declined as KTBA increased. These relationships were not very strong, suggesting only a weak, negative effect of density dependence. The mean \( \pm \text{SE} \) number of
pouch young produced by females in each season (2.6 ± 0.1 in spring, 1.8 ± 0.1 in summer, 1.8 ± 0.1 in autumn and 2.1 ± 0.1 in winter) was comparable to other studies. Dufty (1994c) reported a mean ± SE of 2.2 ± 0.1 in winter in the Hamilton wild population, Heinsohn (1966) 2.3 in all seasons in Tasmania, and Jenkins (1998) 2.0 ± 0.03 (± SD) across all seasons at Woodlands Historic Park.

As density related factors (e.g. social factors, per capita food availability, and disease) made only a minor contribution to the regulation of fecundity, one or more external factors must also have been influencing this population. Increased air temperature negatively affected fecundity, which was also reported by Jenkins (1998) and Todd et al. (2001). Todd et al. (2001) also suggested a relationship between fecundity and rainfall for eastern barred bandicoots at Woodlands Historic Park, but this study showed no relationship between the two. Temperature with a 2-month lag had a stronger negative correlation with fecundity than temperature with a 1-month lag, most likely due to the date of conception being more than one month prior to capture. An increase in fecundity with cooler temperatures may be due to decreased day length, which is highly correlated with temperature (de Lima and Roper 2009). For the nocturnal eastern barred bandicoot, long winter nights could permit increased foraging time and therefore food obtained. This is equivalent to extended feeding times or increased food availability during long days for diurnal species such as the lesser sundas zebra finch Taeniopygia guttata guttata (Perfito et al. 2008) and the desert gerbillid, Sundevall's jird Meriones crassus (Khokhlova et al. 2000). This increase in foraging time would permit the high energy demands of reproduction to be met, resulting in the higher fecundity and larger litter sizes observed in winter and spring. An alternative explanation may be that high energy food sources, such as beetle larvae (Redford and Dorea 1984) are more accessible in winter and spring because they are closer to the soil surface, compared to warmer months when they move down the soil profile in response to high soil temperatures and low soil moisture (Staley et al. 2007). These findings are in agreement with Reimer and Hindell (1996) who concluded that the cooler months in Tasmania, Australia had the most abundant and accessible food sources for eastern barred bandicoots, which allowed the increased energy demands of lactation to be met.

The capture rate of individuals was low at Mooramong, with 50% being captured less than three times and less than 1% captured more than ten times. This could be due to a sampling design that covered only a small area of Mooramong each time and was not replicated between monitoring sessions. In the absence of sampling randomisation and
replication there is no statistically reliable way in which inferences can be made about survival of this population (Williams et al. 2002). Therefore survival estimates could not be determined. However, two phenomena were evident at Mooramong. Firstly, the disappearance rate of adults was high (Fig 4.6), which could suggest high mortality, but is confounded by the trapping grid and trap placement not being consistent over monitoring sessions. Secondly, juvenile disappearance rate could also be high, as 528 pouch young were observed at Mooramong, but only 297 individuals captured. Therefore, at least 57% of juveniles were never captured. This may be a result of mortality, dispersal or eluding traps, but it is not possible to distinguish between the three using the data available.

Mortality rates for juveniles and adults have been found to be highly correlated in mammals, but juvenile mortality is usually higher and more variable than adult mortality (Kraus et al. 2005). High juvenile mortality in eastern barred bandicoots has been reported at other sites as well (Dufty 1991b). In addition, dispersing bandicoots are unlikely to survive long, due to a lack of fox control outside of Mooramong.

Low bandicoot survival is probably due to food shortages or habitat disturbance, including predation by red foxes, exacerbated by drought. The most compelling evidence for this comes from the drought in the mid 1990s. Prior to this drought, most bandicoots were captured in the Nature Reserve, but a gradual shift occurred to the Cottage Dam and then the Homestead Gardens. However, there is very little evidence for individual bandicoots relocating from the Nature Reserve to the Homestead Gardens (Fig 4.5). From April 2001 the number of captures per 100 trap nights declined rapidly around the Homestead and became zero in the Nature Reserve (Fig 4.4). Drought can negatively affect terrestrial grassland invertebrate populations by reducing the quality of host plants (Branson 2008) and by causing soil invertebrates to move deeper down the soil profile (Staley et al. 2007), making them inaccessible to bandicoots. This might result in bandicoots increasing their search effort for food items, leaving them more susceptible to, or less vigilant for predators, but without data on invertebrate abundance it is unknown if food was limited. There was also no data collected on the vegetation structure, but grass biomass is known to be negatively affected by rainfall (Saleem et al. 2009), and habitat disturbance is thought to reduce the ability of prey to escape from predators (Sinclair and Krebs 2002). A decline in grass cover could then have resulted in bandicoots being more vulnerable to red fox predation due to a lack of refuge. The last wild population of bandicoots located in Hamilton, Victoria, Australia was also driven to extinction apparently by the clearing of
undergrowth which reduced the amount of cover and refuge from red foxes (Arnold et al. 1990).

Conclusion

Mooramong is an open reserve. Despite vigorous efforts to control red foxes, these predators could not be completely removed. There was some evidence of negative density dependent regulation on fecundity but temperature made a much stronger contribution to fecundity. The population growth rate showed positive density dependence (demographic Allee effect), but this was not due to a component Allee effect on fecundity. As survival and mortality could not be determined I can only infer that the relationship observed in Fig 4.7 is due to the effect of introduced predators on this population, which are known to cause Allee effects (Courchamp et al. 1999). If this is the case, the only way to reverse this effect is to remove or dramatically reduce introduced predators (i.e. red foxes) in this system (Deredec and Courchamp 2007).
Chapter 5

Fenced in: dynamics of a newly reintroduced population of the critically endangered eastern barred bandicoot *Perameles gunnii*

*The predator-barrier fence surrounding the Hamilton Community Parklands, February 2008*
Chapter 5

Fenced in: dynamics of a newly reintroduced population of the critically endangered eastern barred bandicoot *Perameles gunnii*

Abstract

Understanding how vital rates change and assessing population viability of reintroduced populations should form a critical part of reintroduction management. In reserves surrounded by predator-barrier fences, which restrict dispersal of the endangered species, it is particularly important that the regulatory processes acting upon the population, whether density dependent or independent, are known, so that viability of the population can be predicted and appropriate management decisions made. A population of eastern barred bandicoots was reintroduced into the Hamilton Community Parklands, a 100-ha reserve surrounded by a predator-barrier fence, in July 2007. I monitored this population, using standard cage traps, from July 2007 to February 2010. I collected data on individual bandicoots, including adult weight and sex, and the presence and number of pouch young. Air temperature negatively affected fecundity, with most females breeding and the largest litter sizes found in the cooler months. This could be a response to increased foraging times or an increase in the availability of high energy food sources. There was some evidence of an Allee effect. If this population is suffering from a true Allee effect, then it is likely that Hamilton is too small and will need to be managed as part of a metapopulation.
Introduction

Wildlife reintroductions provide a valuable opportunity to study density dependence because changes in vital rates can be determined as the population grows (Armstrong et al. 2005), provided that the original factors responsible for the extirpation of the species have been removed (Armstrong and Ewen 2002). However, the traditional focus of reintroduction biology has been on determining whether they are successful (Armstrong and Seddon 2008), whereas quantitative demographic analyses of reintroduced populations are scarce (Schaub et al. 2009). Armstrong et al. (2005) argue that assessing the viability of reintroduced populations by understanding how vital rates change with density should form a critical component of all reintroduction programs.

Estimating population parameters, such as reproductive success and survival are the first steps towards understanding population dynamics (de Lima and Roper 2009). For the long term persistance of any population, mortality must be balanced by fecundity (Kraus et al. 2005). Density dependent regulation has been found in populations of the New Zealand robin Petroica australis and saddleback Philesturnus carunculatus (Armstrong et al. 2002), spiny pocket mouse Heteromys desmarestianus (Klinger 2007), prairie vole Microtus ochrogaster and cotton rat Sigmodon hispidus (Reed and Slade 2008). Density related factors can regulate population growth by delaying reproduction, reducing litter size (Reed and Slade 2008) or lowering survival by increasing competition for food (Bonenfant et al. 2009). Density dependence in reintroduced populations is not only important for predicting viability, but also for making management decisions (Armstrong and Ewen 2002). If survival or fecundity is constrained at high density, a population can be harvested with no cost in terms of viability (Armstrong and Ewen 2002).

Density independent factors, such as temperature, precipitation or habitat disturbance can also regulate population growth (Previtali et al. 2009). Rainfall in particular can have strong influences on vital rates of mammals and birds by altering the availability of food, as has been found in the song sparrow Melospiza melodia (Chase et al. 2005), swamp antechinus Antechinus minimus (Sale et al. 2008), European rabbit Oryctolagus cuniculus and house mouse Mus domesticus (Sinclair and Krebs 2002).

Reintroductions usually involve endangered species, which typically have only a limited number of animals available for release. Reintroductions are therefore confronted by different dynamic problems that are typical of small populations, or populations at low
density, one of which is the Allee effect (Deredec and Courchamp 2007). The benefits of conspecific presence can include, but is not limited to: predator dilution or saturation, increased availability of mates, increased fertilisation success, conspecific enhancement of reproduction and reduction of inbreeding, genetic drift, or loss of integrity by hybridisation (Stephens et al. 1999). Therefore, at low population size or density, individual fitness is reduced. This is the Allee effect, which results in a decrease of the population growth rate (Deredec and Courchamp 2007). The ‘component Allee effect’ is a positive relationship between any component of individual fitness and either number or density of conspecifics, the ‘demographic Allee effect’ is a positive relationship between density or population size and per capita growth rate, the latter being the result of the sum of every component Allee effect and of negative density-dependence (Stephens et al. 1999).

The eastern barred bandicoot is critically endangered (Victorian DSE 2007) in Victoria, Australia due to >99% habitat loss and predation by introduced red foxes Vulpes vulpes and cats Felis catus (Backhouse et al. 1994). It is a small (<1kg), nocturnal, solitary marsupial that has an omnivorous diet including beetles, worms and plant matter, such as bulbs of onion grass Romulea rosea and fruit (Winnard and Coulson 2008). It is capable of breeding throughout the year and can produce up to five litters a year, containing an average of two to three pouch young in each (Backhouse et al. 1994). A gestation of just 12.5 days and weaning at 55 days contributes to this fast reproduction rate (Winnard and Coulson 2008). Bandicoots live for 2 - 3 years (Backhouse et al. 1994) and can reach average densities of 1.5 individuals per ha\(^1\) (Minta et al. 1990). Eastern barred bandicoots now exist only in reintroduced populations; eight reintroduction sites have been established in western Victoria since 1989 with varying success (Winnard and Coulson 2008). Today only three continue to have a known population.

In 2007, eastern barred bandicoots were reintroduced into Hamilton Community Parklands, in Western Victoria, Australia. Eastern barred bandicoots had been reintroduced into this reserve in 1989 but became undetectable in 2004 due to an inefficient predator-barrier fence (Winnard and Coulson 2008). In 2005 a ‘floppy top’ was added to the fence (Winnard and Coulson 2008), preventing most introduced predators from gaining access and eastern barred bandicoots from dispersing. I aimed to determine if density dependent or independent mechanisms were responsible for the population rate of change, and if the population could be safely harvested for stocking of other reintroduction sites.
Methods

Study Site

The Hamilton Community Parklands is a 100-ha reserve surrounded by a predator-barrier fence and situated on the outskirts of the city of Hamilton, ~280 km west of Melbourne, Victoria, Australia. A detailed description of the study area has been given in Chapter 3. The long term mean annual rainfall in Hamilton is 613 mm and temperatures range from an annual mean minimum of 7.6°C and an annual mean maximum of 19.0°C. For the study period, mean monthly rainfall was 48 mm, mean monthly maximum temperature 19.6°C and mean monthly minimum temperature 7.6°C. Climate data were obtained from the Bureau of Meteorology (www.bom.gov.au) for the Hamilton airport station, situated ~8.77 km from the Hamilton Community Parklands.

A total of 30 bandicoots were released into the reserve in 2007: 24 (8 male and 16 female) in July 2007 and 6 (4 male and 2 female) in November 2007. No other bandicoots were present in the reserve prior to the initial release. I commenced monitoring the population two weeks after the initial release using standard cage traps, 50 x 18 x 20 cm (length x width x height), baited with peanut butter, honey and rolled oats. I placed two traps on each intersection of a 100 x 150-m permanent grid. From July 2007 to December 2007 I trapped every 39 ± 17.3 (mean ± SE) days, and from December 2007 to February 2010, I increased the trapping interval to 88 ± 9.9 days. Each monitoring session lasted for two or three days, depending on the weather; I did not trap if extreme heat or storms were forecast. When I captured a bandicoot I marked it with a Trovan PIT (passive integrated transponder) tag, and recorded the weight and sex, as well as the number and head length of any pouch young, and the number of lactating teats in adult females. In summer 2008, small patches of the reserve were slashed and burned to create fire breaks, and in summer 2009 only slashing occurred.

Despite the presence of a predator-barrier fence, one cat was known to have entered the reserve and was shot in January 2008. A red fox also entered the reserve in August 2009, but was quickly detected and poisoned. Native predators, such as wedge-tailed eagles Aquila audax, tiger snakes Notechis scutatus and brown snakes Pseudonaja textilis, are present in the reserve but are not controlled.
Data Analysis

I excluded data collected between July 2007 and November 2007 from all statistical analyses because this data was confounded by the release of new animals, which would not have had time to reproduce at the time of monitoring. In total I captured 117 bandicoots (61 male and 56 female) in the reserve, in 581 captures. In addition to this, two bandicoots were found dead in traps and another four escaped whilst handling, before any data could be recorded for them; I did not include any of these animals in any figures or statistical analyses.

All data presentation and statistical analyses follow the same procedure as documented in Chapter 4. Therefore only a summary of those procedures follows. I ran a Kolmogorov-Smirnov two-sample test in GenStat 12 to test for a difference in disappearance rate of males and females. I used the minimum number of bandicoots known to be alive (KTBA) for consistency with Chapter 4 and because mark-recapture estimates were unsuitable, as some trap sessions could be conducted for only two consecutive nights. Therefore, despite the limitations of KTBA (Jolly and Dickson 1983, Nichols and Pollock 1983, Pollock et al. 1990), it provides a lower limit of population size (Jolly and Dickson 1983). I caught lactating females on 19 occasions of which four females had pouch young as well as young in a nest. I only included these females in the breeding generalised linear mixed model (GLMM) and not in any other analysis. I calculated the minimum adult female breeding weight by taking the mean female breeding weight (771 g) and subtracting 2 standard deviations (232 g). Two females with pouch young were below this weight. Therefore I excluded them from all analyses. I ran all KTBA and rate of increase regressions in SPSS 17.0. I fitted two GLMMs in GenStat 12 to examine the effect of mean monthly temperature and mean monthly rainfall on (i) litter size, using a Poisson distribution and (ii) breeding using a binominal distribution. Both models were set up the same way as in Chapter 4 with temperature and rainfall being lagged by 1, 2 and 3 months. Individual females and trap session were random factors in both models. In the litter size model none of the climatic variables was significant, either when placed in the model alone or when combined with another variable. In the breeding model all temperature variables were significant when placed in the model alone, but only temperature with a 1-month lag retained its significance when combined with temperature with a 2 and 3-month lag.
Results

Since monitoring began, this population has experienced more population increases than declines, with 58% of monitoring sessions showing an increase, 17% no change and 25% a population decline (Fig 5.1). Once all 30 founder bandicoots had been released, the population remained around 30-35 KTBA for the majority of trapping occasions (Fig 5.2). In spring 2008 the population increased to 51 KTBA but had declined to 35 KTBA by the next monitoring session in summer 2009. The population then increased again to 50 KTBA in winter 2009 and remained at that level in spring 2009 before returning to 30 KTBA in summer 2010 (Fig 5.2). Only two founder bandicoots were not captured after release.

Fig 5.1: Population rate of increase of eastern barred bandicoots at Hamilton Community Parklands from October 2007 to Feb 2010. Bandicoots were released in July 2007 (n = 24) and November 2007 (n = 6).
Fig 5.2: Number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from July 2007 to February 2010. Bandicoots were released in July 2007 ($n = 24$) and November 2007 ($n = 6$).

Of the 117 bandicoots caught during this study, 50% were captured two or fewer times and only 10% were captured 10 or more times. There was no significant difference in the capture rates of males ($\bar{x} = 4.21$, $SD = 2.18$) and females ($\bar{x} = 4.71$, $SD = 2.60$; $t = 0.91$, $df = 115$, $P = 0.273$). The low capture rate was coupled with a fast disappearance rate. Three bandicoots (1 male and 2 female) were captured in the reserve for 28 months (Fig 5.3), all being founders from the first release in July 2007. Although the actual age of founder bandicoots was known, Fig 5.3 shows only the number of months over which they were captured, and not how long they were known to live, for consistency with wild-bred bandicoots. Half of all males ($n = 31$) and females ($n = 29$) disappeared after four and seven months respectively ($\chi^2 = 3.65$, $df = 2$, $P = 0.161$). In addition to bandicoot captures, in each monitoring session an average of 42% ($8 - 68\%$) of traps had been triggered and were closed at the time of checking. These traps were either closed with bait present, closed without bait, or contained a non-target species, mostly common brushtail possums *Trichosurus vulpecula*. 
In total, 248 pouch young in 107 litters were recorded during this study, comprising 13 litters with one pouch young, 48 with two, 45 with three and one litter of four. This did not include the litters carried by two females classed as juveniles; one of these had one pouch young and the other had two, one of which was dead upon capture. The percentage of females with pouch young fluctuated between 0% and 100% (Fig 5.4). The highest mean litter size was 2.9 in October 2007 (Fig 5.5), when all bandicoots \((n = 14)\) had pouch young. Zero pouch young were recorded on three occasions, in January 2009, March 2009 and February 2010 (Fig 5.5). Females had a seasonal mean ± SE of 2.5 ± 0.1 \((n = 52)\) pouch young per litter in spring, 2.0 ± 0.1 \((n = 21)\) in summer, 1.8 ± 0.2 \((n = 6)\) in autumn and 2.5 ± 0.1 \((n = 27)\) in winter.

**Fig 5.3:** Cumulative distribution frequency of the number of months male and female eastern barred bandicoots were known to be alive at Hamilton Community Parklands between July 2007 and February 2010.
Chapter 5: Dynamics of reintroduction

Fig 5.4: Proportion of female eastern barred bandicoots with pouch young at Hamilton Community Parklands from July 2007 to February 2010. In January 2009, March 2009 and February 2010 all females had zero pouch young.

Fig 5.5: Mean litter size of female eastern barred bandicoots at Hamilton Community Parklands from July 2007 to February 2010. In January 2009, March 2009 and February 2010 all females had zero pouch young.
There was a significant positive relationship between KTBA and rate of increase ($r^2 = +0.45$, $df = 9$, $P = 0.035$; Fig 5.6). There was also a positive but non significant relationship between rate of increase and the proportion of females with pouch young ($r^2 = +0.15$, $df = 9$, $P = 0.274$; Fig 5.7) and mean litter size ($r^2 = +0.44$, $df = 9$, $P = 0.105$; Fig 5.8). Mean monthly precipitation with a 1, 2 and 3-month lag had no significant effect on the population rate of increase ($r^2 = +0.012$, +0.098 and +0.223 respectively). Mean monthly temperature with a 1, 2 and 3-month lag also made no significant contribution to rate of increase ($r^2 = +0.310$, +0.270 and +0.357 respectively). Combining rainfall and temperature data for each monthly lag did not increase the amount of explained variance, or cause the data to reach statistical significance, nor did combining the three rainfall or temperature variables in the one model.

![Fig 5.6: Linear regression of the population rate of increase on the number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from December 2007 to February 2010.](image-url)
Fig 5.7: Linear regression of the proportion of females with pouch young on the number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from December 2007 to February 2010.

Fig 5.8: Linear regression of mean litter size on the number of eastern barred bandicoots known to be alive (KTBA) at Hamilton Community Parklands from December 2007 to February 2010.
In the final models (GLMMs), litter size was not correlated with any temperature or rainfall variable, whereas breeding was negatively correlated with temperature with a 1-month lag \((P = 0.027)\). In this model, the odds ratio reflects the strength of the correlation: for each additional degree of temperature with a 1-month lag, the log odds of breeding is predicted to decrease by a factor of \(e^{-0.530} = 0.59\) (95% confidence intervals: 0.40, 0.87) or 41%.

**Discussion**

Within 31 months of reintroduction, the Hamilton Community Parklands population has experienced more population growth than declines (Fig 5.1), and appears to have oscillated at around 33 known to be alive (KTBA), with numbers peaking to ~51 KTBA in spring 2008 and winter and spring 2009 (Fig 5.2). These peaks indicate that, during part of the year, fecundity is exceeding mortality until what appears to be the carrying capacity of the reserve is reached, but at other times mortality exceeds fecundity.

The seasonal pulses observed in KTBA in spring 2008 and winter and spring 2009 can be explained by the increase in breeding and litter size in the cooler months (Fig 5.4 & 5.5), but a lag between the actual and observed increase in KTBA could explain why a peak was not observed in winter 2008. The majority of bandicoots were captured as adults, and would have been at least 4 - 6 months old (Heinsohn 1966) upon first capture. It is possible that the actual lag between birth and first capture is one year. Therefore any breeding that occurred in winter 2007 would have been detected only in late winter 2008, after the winter 2008 monitoring. The increase in population size in November 2007 is largely attributable to the release of six bandicoots earlier that month and not an increase in fecundity.

In total, 117 bandicoots were caught, including 28 founders, and 248 pouch young were observed at Hamilton. Females had a seasonal mean ± SE of 2.5 ± 0.1 pouch young per litter in spring, 2.0 ± 0.1 in summer, 1.8 ± 0.2 in autumn and 2.5 ± 0.1 in winter, which is comparable to the litter sizes at Mooramong (spring: 2.6 ± 0.1, summer: 1.8 ± 0.1, autumn: 1.8 ± 0.1 and winter: 2.1 ± 0.1; Chapter 4) and other studies conducted at different locations. Dufty (1994c) reported a mean ± SE of 2.2 ± 0.1 in the Hamilton wild population, Heinsohn (1966) 2.3 in Tasmania, and Jenkins (1998) 2.0 ± 0.03 (± SD) at Woodlands Historic Park.
The capture rate of individuals was low at Hamilton, with 50% being captured two or fewer times and only 10% captured more than ten times. This could have been due to high trap interference by non-target species, particularly common brushtail possums. Noticeable declines in the KTBA were observed (Fig 5.2), which must have been due to mortality, either through interspecific competition, declining resources, such as food or nest sites, or increases in predation by native predators. This observed mortality must have been mainly of adults, as the trapping record was predominantly made up of adults. Juvenile mortality may also have been high, as a total of 248 pouch young were observed and only 117 individuals captured, but this may have been a consequence of a lag between birth and first capture. Despite the disappearance rate being high (Fig 5.3), no bandicoots could be presumed dead, as this reintroduction was only recent and had not yet exceeded the life expectancy of a bandicoot (2 - 3 years) (Backhouse et al. 1994). Some bandicoots had also eluded traps for up to 16 months. Therefore capture and disappearance rates may be expected to change with future monitoring data.

A demographic Allee effect is indicated by the data: when KTBA was low, population growth rate ($r$) was also low (Fig 5.6) and $r$ was negative below a KTBA of ~40. There was also some indication of a component Allee effect on fecundity: as KTBA declined so too did the proportion of females with pouch young (Fig 5.7) and their mean litter size (Fig 5.8). However, these relationships were not statistically significant and the observed patterns may be a result of a small data set. Allee effects are notoriously difficult to determine (Courchamp et al. 1999) and more data must be collected on this population before any inferences are made. However, if this is a true Allee effect it is unlikely to be caused by an inability to find a mate as this population is thought to have reached carrying capacity. Instead, some other aspect of bandicoot reproduction may be involved, such as the influence of sexual selection or genetic distance between partners (Deredec and Courchamp 2007).

Despite some evidence of an Allee effect on fecundity, external factors also regulated fecundity. As found at Mooramong (Chapter 4), there was a strong negative correlation between temperature and breeding at Hamilton, but no correlation was found between litter size and temperature or temperature and rate of increase. This could be due to low sample size, which could also explain why temperature with a 1-month lag was more strongly correlated with breeding at Hamilton, rather than the 2-month lag more strongly correlated with breeding and litter size at Mooramong (Chapter 4). Day length is highly
correlated with temperature (de Lima and Roper 2009). Therefore, during the cooler months, the longer nights would permit the nocturnal eastern barred bandicoot to forage for longer periods. This could lead to an increase in food consumption and nutritional quality, particularly as high energy food items such as beetle larvae (Redford and Dorea 1984) are more accessible in cooler weather, as soil invertebrates tend to be closer to the soil surface (Staley et al. 2007). This increase in food availability or nutritional quality of food could permit an increase in fecundity as seen in other species, such as the black-throated blue warbler *Dendroica caerulescens* (Nagy and Holmes 2005) and the short-tailed shrew *Blarina brevicauda* (Bronson 2009).

Only one red fox and one cat are known to have gained access to the reserve during this study. The red fox entered the reserve between the winter 2009 and spring 2009 monitoring sessions and no noticeable decline in KTBA was observed. Some habitat disturbance occurred but appeared to contribute little to mortality in this population. In summer 2008, small patches of the reserve were slashed and burned to create fire breaks; slashing also occurred in summer 2009. Although no deaths were reported during these events, it is likely that they had some impact on a small number of bandicoots. Therefore, mortality due to external factors cannot be discarded.

**Conclusion**

There was no support for the Hamilton population being regulated by negative density dependent processes, but some evidence of a demographic Allee effect. If this is a true Allee effect then releasing more bandicoots into Hamilton would be the most effective way in which to counteract the consequences of these effects (Deredec and Courchamp 2007). However, given the small size (100 ha) of Hamilton Community Parklands, and that it is surrounded by a predator-barrier fence preventing bandicoot dispersal, it is possible that this reserve has reached carrying capacity. Further releases would therefore not change the observed pattern, suggesting that Hamilton Community Parklands is too small to hold a viable, self-sustaining population and may need to be managed as part of a metapopulation. If so, this population would not be suitable for harvesting to stock other release sites.
Chapter 6

The foraging habitat use and movements of reintroduced eastern barred bandicoots *Perameles gunnii* into an unoccupied reserve

*Grassy woodland habitat at Hamilton Community Parklands, July 2007*
Chapter 6

The foraging habitat use and movements of reintroduced eastern barred bandicoots *Perameles gunnii* into an unoccupied reserve

Abstract

Understanding how the distribution of resources affects the spatial occupation and movements of a reintroduced population over time is paramount for the successful establishment and persistence of the population in question. In this study I determined foraging habitat use, at the population scale, of eastern barred bandicoots *Perameles gunnii* reintroduced into Hamilton Community Parklands in 2007. This reserve is surrounded by a predator-barrier fence and regularly monitored for incursions by the red fox *Vulpes vulpes*, the main introduced predator responsible for the demise of this species. I determined foraging habitat use by counting foraging digs on a permanent grid established across the reserve. I fitted linear mixed models to determine which measured habitat variables were associated with foraging digs in two phases: the reintroduction and established phase. Soil compaction was the only significant variable in the reintroduction phase, whilst in the established phase digs varied seasonally with soil compaction and with grass cover and distance to cover for all seasons combined. In the second part of this study I investigated spatial occupation by founder and wild-bred bandicoots based on seasonal trapping data. Males of both origin occupied larger areas than females, and founder bandicoots occupied larger areas than wild-bred bandicoots. The smaller areas occupied by wild-bred bandicoots can be interpreted as a trade-off between increasing male-female interactions and decreasing male-male aggressive interactions in a high-density population. The larger areas occupied by founders is probably due to males attempting to increase mating opportunities with patchily-distributed females in the reintroduction phase, then retaining their larger ranges in the established phase by being dominant over younger, wild-bred bandicoots.
Introduction

The loss of suitable habitat has been a major factor contributing to the decline and extinction of many animals worldwide (Michel et al. 2008). For this reason, recent conservation biology programs have focused on restoring ecosystems and preventing further extinctions by reintroducing threatened species (Michel et al. 2008, IUCN 1998). Re-establishing species in areas where they formerly occurred must consider the suitability of habitats at proposed release sites (Wolf et al. 1998, IUCN 1998). Determining habitat suitability requires an understanding of the species’ habitat preferences, which can be provided by quantitative exploration of its resource use patterns (Oppel et al. 2004).

Documenting space use and movement patterns, and their influence on survival and reproduction, can also help to evaluate the progress of reintroduction projects, because they are important indicators of successful adaptation of reintroduced animals to the new environment (Benson and Chamberlain 2007).

Habitat selection studies aim to understand where animals are more likely to occur based on resource availability and patterns of use (Edge et al. 2010). Although often difficult to determine for translocated species, the habitat quality and quantity of the release site are critical to the outcome of translocations (Wolf et al. 1996, Armstrong and Seddon 2008). Presumably, a habitat type is selected because it provides the resources that allow the animal to meet a biological or physiological need (Edge et al. 2010). Habitats must provide individuals with resources that best satisfy their requirements, including adequate food, shelter, escape from predators and access to mates (Michel et al. 2008, Preatoni et al. 2005). Identifying the patterns of habitat selection is complicated by the inherent plasticity in habitat selection, which is a function of population density, resource availability and species interactions (Rodhouse et al. 2010). The biological needs of animals can also change across seasons, further complicating habitat selection studies (Edge et al. 2010).

Costs associated with increasing population density include an increase in mortality due to resource depletion, as seen in aggregating species such as the processionary caterpillar Ochrogaster lunifer (Floater 2001) and an increase in the rate of aggressive interactions, which reduces the overall quality of the habitat, as seen in Allenby’s gerbil Gerbillus andersoni allenbyi (Ovadia and Dohna 2003). Two related theories predicting how individuals will occupy habitat as density increases are the ideal free distribution and the ideal despotic distribution. Ideal free distribution theory assumes that individuals are free
to occupy the habitat of their choice so that the mean fitness in each habitat is equal (Morris 2003b). Borgo (2003) reported that reintroduced Alpine marmots *Marmota marmota* conformed to an ideal free distribution. The ideal despotic distribution predicts that immigrants will settle according to the distribution of resources, and the density and rank of resident individuals (Murray *et al.* 2007). Therefore, an individual will perceive that the value of each habitat is reduced by interference from dominant occupants (Morris 2003b). This has been observed in white-footed mice *Peromyscus leucopus*, where territorial breeding females excluded subordinate animals from the highest-quality habitat (Morris 2003a). With both theories, mean individual fitness declines as a function of resident density (Greene and Stamps 2001) and patches will be chosen based on cost/benefit analysis of the respective density qualities of those patches (Morris 2003a).

Reintroduced individuals are usually raised in captivity and therefore naive to the natural environment (Banks *et al.* 2002). This naiveté is thought to increase their short term vulnerability to predators after release (Banks *et al.* 2002) because they are forced to find resources in an unfamiliar environment (Moehrensclager and Macdonald 2003). Animals unfamiliar with a reintroduction site typically move very little soon after release, presumably because they are unfamiliar with safe sources of food and escape routes from predators (Banks *et al.* 2002). This response has been observed in wild-bred red squirrels *Sciurus vulgaris* (Wauters *et al.* 1997) and captive-bred dormice *Muscardinus avellanarius* (Bright and Morris 1994). Alternatively, some species such as translocated wild-bred water voles *Aruicola terrestris* and brown bears *Ursus arctos* make extensive post-release movements (Moorhouse *et al.* 2009, Preatoni *et al.* 2005).

The eastern barred bandicoot is a critically endangered (Victorian DSE 2007) species that now only survives in reintroduced populations in Victoria, Australia (Winnard and Coulson 2008). This bandicoot is thought to have declined mainly due to predation by introduced predators, particularly the red fox *Vulpes vulpes* and domestic cat *Felis catus*, and habitat loss; >99% of Victoria’s native grasslands and grassy woodlands have been lost through habitat alteration and destruction (Backhouse *et al.* 1994). This grassy woodland species is known to nest in areas of high structural complexity during the day and forage in open grasslands, on a variety of invertebrates and plant matter at night (Heinsohn 1966). It is short-lived, with an average life span of just 2 - 3 years, but is highly fecund (Backhouse and Crosthwaite 1996). Females can reproduce throughout the year and are capable of producing up to five litters per year, with an average of two to three young in each litter.
The most common social behaviour observed by Dufty (1994a) was aggressive defence of foraging resources, but in general bandicoots tend to avoid intraspecific interactions suggesting that the ideal despotic distribution could apply to their populations.

I investigated habitat use, at the population scale, of eastern barred bandicoots reintroduced into a fox-free reserve. The aim of this study was to determine how habitat use changed as the population became established, by comparing habitat variables and foraging digs between the initial reintroduction phase and the established phase. I also aimed to determine if there was a difference in spatial occupation and movements of founders captured in the reintroduction phase compared to those captured in the transition and established phases combined. Finally I aimed to determine if there was a difference between the spatial occupation and movements of founders and wild-bred bandicoots. The results of this study are necessary to refine reserve management and release protocols, which will enable the long-term viability of this species.

**Methods**

**Study Site**

The Hamilton Community Parklands is a 100-ha grassy woodland reserve surrounded by a predator-barrier fence and monitored regularly for any red fox incursions. It is situated on the outskirts of the city of Hamilton which is 280 km south west of Melbourne, Victoria, Australia. A full description of the reserve has been given in other chapters. This reserve had previously been used for an eastern barred bandicoot reintroduction in 1989 but by 2004 had failed due to zero bandicoots being captured (Winnard and Coulson 2008).

In July 2007, after an upgrade of the predator-barrier fence, 24 (8 male and 16 female) bandicoots were released into the reserve. These bandicoots were released in four equal groups into woodland areas (Fig 6.1). A further six (4 male and 2 female) bandicoots were released in November 2007, this time in two groups in the open grasslands approximately 100 m from the woodland edge (Fig 6.1). Two weeks after the first release, I trapped the population using standard cage traps baited with peanut butter, honey and oats. I placed two traps at each intersection of a 100 x 150-m grid (n = 60) that covered the entire reserve. This trapping occurred over two or three days depending on weather conditions,
and was repeated seasonally for 2.5 years (July 2007 to February 2010). When I caught a bandicoot, I used the methods detailed in Chapter 3 to process the bandicoot. I recorded the sex, weight, and any visible parasites and injuries. I also checked females for pouch young and recorded the number and size of young, or any signs of lactation indicative of young left in a nest.

During the monitoring period I also conducted foraging dig counts and habitat measurements in plots at each intersection of the established grid, in each season between July 2007 and autumn 2009. I counted foraging digs within a 5-m radius from the centre of the plot. I assumed that digs were recently made, as small disturbances to the soil readily changed their shape. From winter 2007 to autumn 2008, I measured habitat in plots with a radius of 10 m. Within these plots I visually estimated the percentage cover of grass below and above 15 cm (the average height of a bandicoot), woody debris (twigs and sticks), litter (grass or leaves), moss, rock and water. I also counted the number of trees and bushes within the plot, the size of any coarse woody debris (CWD; one or more pieces of woody material where at least one piece was ≥0.5 m long) and the distance to cover, which was
defined as any structure that could conceal a bandicoot, such as trees and CWD. I took a photograph, using a Fujifilm FinePix S5500 digital camera, from the centre of the plot, 15 cm above the ground looking up, which was used to calculate the projective foliage cover (PFC). I also took photographs of a 15 x 25-cm red board (the approximate height and length of a bandicoot) from a distance of 10 m and a height of 40 cm (the approximate height of a red fox), at each of the four cardinal directions. These photographs were used to determine the percentage cover available to conceal a bandicoot. I analysed all photographs in Adobe Photoshop CS. I measured soil compaction at the centre of the plot and at each of the four cardinal directions 10 m from the plot centre using a pocket penetrometer (Geotest Instrument Corp). I then took a 10-cm deep soil core, 1 m from the centre of the plot, which I sieved to remove any vegetation and invertebrates. I transferred invertebrates and bulbs of onion grass *Romulea rosea* to a 70% alcohol solution and dried the soil at 105°C for 24 h (Australian Standard AS 1289.2.1-2005) to determine soil moisture content. I placed a pitfall trap (a plastic drinking cup, 7 cm in diameter) containing ~70 ml of a 70:30 solution of ethanol and glycerol at each plot, leaving them open for three days and nights, and transferring invertebrates to 70% ethanol. I later identified all invertebrates to order level, and then dried them at room temperature for a minimum of one week to determine dry weight biomass. Only the biomass of food items known to be eaten by eastern barred bandicoots was used in the data analysis. Variables measured were chosen because they were considered important features of bandicoot habitat in previous studies (Cook 2001, Dufty 1991a, Dufty 1991b, Dufty 1994b, Ferguson 2006, Minta *et al.* 1990 and Ch 7).

I repeated these methods for winter 2008 to autumn 2009, with the exception that I used plots that had only a 5-m radius due to the dense grass cover and a large number of plots returning 100% cover using the cover board method. I did not collect onion grass bulbs or soil invertebrates during these seasons due to very low sample sizes in the previous four seasons.

**Data analysis**

I first calculated the mean number of digs in grassland and grassy woodland areas in each of the eight seasons. Any plots that contained zero digs were removed from all analyses to prevent biased detection error, which is an inherent problem in habitat selection studies, particularly when cryptic species such as small mammals are involved (Rodhouse *et al.*
In this study I excluded plots containing zero digs for two reasons: i) non-detection of the species is not equivalent to absence, because a species can go undetected when present (MacKenzie et al. 2002, Tyre et al. 2003) and ii) as this was a reintroduction, non-detection of digs may not have been due to bandicoots rejecting that area for foraging, but because they had not encountered that area yet. I then used Fisher’s least significant difference post-hoc test in GenStat 12 to determine if there was a significant difference between the number of digs found in grassland and grassy woodland plots. Based on the results from calculating these means and because the habitat analysis methodology varied slightly between the first and second year (10-m radius plot in the first year compared to a 5-m radius plot in the second year), I ran linear mixed models (LMM) in GenStat 12 on i) the initial reintroduction phase, winter 2007 and spring 2007, when bandicoots were not found across the whole reserve and ii) the established phase, winter 2008 to summer 2009, when bandicoots and their digs were found across the reserve. I did not run a model on data collected in summer 2008 and autumn 2008, because these seasons appear to be a transition phase between the reintroduction and establishment phases, based on relative dig counts in grassland and woodland.

I used LMM to determine which habitat variables were associated with foraging digs. Again, I used only plots that contained more than one dig in the models. Digs were loge transformed, to better satisfy the assumptions of constant variance and normality of the errors. In both models, ‘digs’ was the response variable and ‘plot’ was a random factor. Prior to running the models I calculated the correlations between explanatory variables; cover board was correlated with grass (r = +0.54) in the established phase model and soil compaction was correlated with soil moisture (r = +0.56). I then added individual variables to the model sequentially to test for a significant (P ≤0.05) seasonal interaction. There was no seasonal interaction in the reintroduction phase model, and only soil compaction showed a significant seasonal interaction in the established phase model. I then added all variables to each model using a backwards elimination approach, and deleted any variable that did not make a significant (P ≤0.05) contribution to the model. The final reintroduction phase model contained only soil compaction; there was no seasonal interaction in this model due to a very narrow spread of the data in winter 2007 and very little overlap of the data for both seasons. The final established phase model contained soil compaction with a seasonal interaction, % cover of grass (both grass height categories were combined as when run in the model individually they were both significant) and distance to cover.
I used two measures of ranging behaviour, range span and 95% minimum convex polygons (MCP), to test whether there was a difference between males and females, and founders and wild-bred bandicoots. I used 95% MCP in Ranges7 for all bandicoots that had been captured a minimum of 5 times. I chose MCP because the low sample sizes of locations per individual were low, and my aim was not to determine home range size per se but rather an approximate measure of spatial coverage for comparative purposes. Two founder male bandicoots had an MCP that lay outside of the fenced area, so I subtracted the area outside the fence from their MCP value. I ran a general ANOVA in GenStat 12 to determine if there was a difference between sex and origin (founders and wild-bred) for MCP, range span, number of captures and number of plots caught at. I log_{e} transformed MCP and span because fitted value plots revealed that the untransformed data were not normally distributed.

To investigate if male and female founder bandicoots occupied larger areas shortly after release than they did later, I ran paired t-tests in SPSS 17.0. I compared the MCP and span of male founder bandicoots caught in the reintroduction phase (winter and spring 2007) with their MCP and span in the transition and established phases combined (summer 2008 to spring 2009); I then repeated this analysis for female founders. I captured no founders in February 2010, so excluded this trap session from the analysis. I used only bandicoots that had been captured in both periods in this analysis. If the data violated the assumption of equal variance, I used the relevant output to compensate for this.

**Results**

In the reintroduction phase (winter 2007 and spring 2007), bandicoot foraging digs were found in 40 - 50% of plots. During the transition phase (summer 2008 and autumn 2008) they were found in 63 - 72% of plots, whilst in the established phase (winter 2008 to autumn 2009) 92 - 97% of plots contained digs. In the reintroduction phase, bandicoots foraged more in grassy woodland patches than grassland patches but there was only a significant difference found in winter 2007 (Fig 6.2). In the established phase, bandicoots switched to foraging more in the open grassland patches but this was not significant in any season (Fig 6.2).
In the reintroduction phase, the only measured variable that foraging bandicoots were selecting for was soil compaction ($P = 0.003$). As this variable decreased, the number of digs increased, but there was no interaction with season. In the established phase, bandicoots were foraging more in plots containing less compact soil, with a higher percentage of grass cover and closer to cover. The interaction between soil compaction and season was significant $P = 0.012$. Therefore different equations were constructed for each season. The seasonal interaction with soil compaction was greatest in winter 2008, with soil compaction negatively related to digs. In spring 2008, summer 2009 and autumn 2009 the same pattern was observed, but the coefficient (slope) for soil compaction (Table 6.1) was very close to zero, indicating that soil compaction was having little effect on digs in those seasons. There was no seasonal interaction for grass cover $P < 0.001$ and distance to cover $P = 0.039$ for all seasons combined.

In the reintroduction phase, the number of digs was predicted to decline by $e^{-0.48} = 0.62$ or 38% for every 1 kg/cm$^2$ (back transformed to the original scale) increase in soil compaction.
Once the bandicoots had become established across the reserve the number of digs was predicted to change by a factor of $e^{-0.93} = 0.39$ or 61% decline for every 1 kg/cm$^2$ increase in soil compaction in winter 2008 only. For all other seasons, the number of digs was predicted to change but at a slower rate; -16% (spring 2008), +3% (summer 2009) and -12% (autumn 2009) for every 1 kg/cm$^2$ increase in soil compaction. For every 10% increase in grass cover, digs were predicted to increase by a factor of $e^{0.02\times10} = 1.25$ or 24%, and digs were predicted to decline by 9% for every 10 m increase from cover.

Table 6.1: Output of the eastern barred bandicoot reintroduction phase (winter and spring 2007) and established phase (winter 2008 to autumn 2009) linear mixed models, showing the estimated coefficients and standard errors (SE) for all significant habitat variables in each season at Hamilton Community Parklands. Values in the table fit into the general equation:

$$\log_e(digs) = a + b \times \text{soil compaction} + c \times \text{grass cover} + d \times \text{distance to cover}$$

<table>
<thead>
<tr>
<th>Phase/ Season</th>
<th>Soil Compaction</th>
<th>% Grass Cover</th>
<th>Distance to Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Intercept (a)</td>
<td>Coefficient (b)</td>
<td>SE (c)</td>
</tr>
<tr>
<td>Reintroduction</td>
<td>1.58</td>
<td>-0.48</td>
<td>0.15</td>
</tr>
<tr>
<td>Winter 2008</td>
<td>1.83</td>
<td>-0.93</td>
<td>0.27</td>
</tr>
<tr>
<td>Spring 2008</td>
<td>2.22</td>
<td>-0.18</td>
<td>0.27</td>
</tr>
<tr>
<td>Summer 2009</td>
<td>1.67</td>
<td>0.03</td>
<td>0.27</td>
</tr>
<tr>
<td>Autumn 2009</td>
<td>2.51</td>
<td>-0.12</td>
<td>0.27</td>
</tr>
</tbody>
</table>

Bandicoots captured in the reintroduction phase had a total of eight trapping occasions to be captured, whereas bandicoots captured in the transition and established phases combined had a total of 26 possible trap occasions. In the reintroduction phase, the mean ± SE maximum distance male founders travelled from the release site was $612.6 \pm 107.5$ m (range 321.2 – 1042.0 m), whereas females moved a maximum mean distance of $218.7 \pm 35.1$ m (range 75.6 – 510.6 m). There was no significant difference between the MCP and range span of male and female founder bandicoots caught in the reintroduction phase compared with their MCP and span in the transition and established phases combined (Table 6.2 & 6.3). However, there was a significant difference between the number of captures (female only) and plots at which male and female bandicoots were
caught. Due to the small sample sizes and non significant results, power analyses were performed in G*Power 3.1. For MCP, the effect size was $d = 1.74$ and power $= 0.92$ for males, and $d = 2.00$ and power $= 0.99$ for females, indicating that the sample size was adequate to detect a large effect. However for range span the effect size was $d = 0.12$ and power $= 0.06$ for males and females, signifying that the sample size was too small, resulting in a possible Type 2 error.

Of the 30 founders released into the reserve, only 28 were re-captured. An additional 89 wild-bred bandicoots were captured. Founders had significantly larger MCPs and range span than wild-bred bandicoots, and were also caught at significantly more plots and captured more times (Table 6.2 & 6.3). Male bandicoots also had significantly larger MCPs and span than females, and were caught at significantly more plots. There was no significant difference between the number of times males and females were captured (Table 6.4). The interaction between sex and origin was not significant for any of the four variables (Table 6.4).

Table 6.2: Descriptive statistics for founder eastern barred bandicoots caught in the reintroduction phase (winter and spring 2007) and transition and established phases combined (summer 2008 to spring 2009) at Hamilton Community Parklands.

<table>
<thead>
<tr>
<th></th>
<th>Reintroduction</th>
<th>Transition &amp; Established</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Male</td>
<td>Female</td>
</tr>
<tr>
<td>MCP 95 (ha)</td>
<td>$mean$ 6.7 ± 1.5</td>
<td>0.3 ± 0.3</td>
</tr>
<tr>
<td></td>
<td>$range$ 1.6 - 11.0</td>
<td>0 - 2.3</td>
</tr>
<tr>
<td></td>
<td>$n$ 6</td>
<td>9</td>
</tr>
<tr>
<td>Span (m)</td>
<td>$mean$ 733.7 ± 103.2</td>
<td>216.7 ± 37.0</td>
</tr>
<tr>
<td></td>
<td>$range$ 449.8 - 1111.0</td>
<td>0 - 321.7</td>
</tr>
<tr>
<td></td>
<td>$n$ 6</td>
<td>9</td>
</tr>
<tr>
<td>Captures</td>
<td>$mean$ 5.0 ± 0.4</td>
<td>2.8 ± 0.4</td>
</tr>
<tr>
<td></td>
<td>$range$ 4 - 6</td>
<td>2 - 5</td>
</tr>
<tr>
<td></td>
<td>$n$ 6</td>
<td>9</td>
</tr>
<tr>
<td>Plots</td>
<td>$mean$ 4.2 ± 0.3</td>
<td>2.1 ± 0.3</td>
</tr>
<tr>
<td></td>
<td>$range$ 3 - 5</td>
<td>1 - 4</td>
</tr>
<tr>
<td></td>
<td>$n$ 6</td>
<td>9</td>
</tr>
</tbody>
</table>
Table 6.3: Results from paired t-tests run in SPSS on 95% minimum convex polygons (MCP), span, capture and plot data collected from founder eastern barred bandicoots caught in the reintroduction phase (winter and spring 2007) and transition and established phases combined (summer 2008 to spring 2009) at Hamilton Community Parklands.

<table>
<thead>
<tr>
<th></th>
<th>t</th>
<th>df</th>
<th>P (2 tailed)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>MCP 95</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>males</td>
<td>-1.15</td>
<td>5</td>
<td>0.302</td>
</tr>
<tr>
<td>females</td>
<td>-1.85</td>
<td>8</td>
<td>0.102</td>
</tr>
<tr>
<td><strong>Span</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>males</td>
<td>-0.12</td>
<td>5</td>
<td>0.909</td>
</tr>
<tr>
<td>females</td>
<td>-0.11</td>
<td>8</td>
<td>0.913</td>
</tr>
<tr>
<td><strong>Captures</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>males</td>
<td>-1.49</td>
<td>5</td>
<td>0.196</td>
</tr>
<tr>
<td>females</td>
<td>-2.77</td>
<td>8</td>
<td>0.024</td>
</tr>
<tr>
<td><strong>Plots</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>males</td>
<td>-3.16</td>
<td>5</td>
<td>0.025</td>
</tr>
<tr>
<td>females</td>
<td>-3.27</td>
<td>8</td>
<td>0.011</td>
</tr>
</tbody>
</table>

Table 6.4: Descriptive statistics, on untransformed data, for 95% minimum convex polygons (MCP), span, number of times an individual eastern barred bandicoot was captured and the number of different plots a bandicoot was captured at Hamilton Community Parklands from winter 2007 to autumn 2009. An MCP of 0.0 is due to an individual being captured at ≤2 plots.

<table>
<thead>
<tr>
<th></th>
<th>Males</th>
<th>Females</th>
<th>Founders</th>
<th>Wild-bred</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>MCP 95 (ha)</strong></td>
<td>mean 7.5 ± 1.4</td>
<td>1.3 ± 0.4</td>
<td>6.0 ± 1.6</td>
<td>3.2 ± 0.7</td>
</tr>
<tr>
<td></td>
<td>range 0 - 23.3</td>
<td>0 - 8.6</td>
<td>0 - 23.3</td>
<td>0 - 10.6</td>
</tr>
<tr>
<td></td>
<td>n 24</td>
<td>21</td>
<td>23</td>
<td>22</td>
</tr>
<tr>
<td><strong>Span (m)</strong></td>
<td>mean 554.7 ± 57.4</td>
<td>216.7 ± 28.0</td>
<td>476.1 ± 65.0</td>
<td>314.2 ± 46.4</td>
</tr>
<tr>
<td></td>
<td>range 102.6 - 1153.5</td>
<td>0 - 500.9</td>
<td>98.5 - 1153.5</td>
<td>0 - 769.7</td>
</tr>
<tr>
<td></td>
<td>n 24</td>
<td>21</td>
<td>23</td>
<td>22</td>
</tr>
<tr>
<td><strong>Captures</strong></td>
<td>mean 9.6 ± 1.1</td>
<td>10.4 ± 1.2</td>
<td>12.0 ± 1.3</td>
<td>7.9 ± 0.7</td>
</tr>
<tr>
<td></td>
<td>range 5 - 27</td>
<td>5 - 22</td>
<td>5 - 27</td>
<td>5 - 15</td>
</tr>
<tr>
<td></td>
<td>n 24</td>
<td>21</td>
<td>23</td>
<td>22</td>
</tr>
<tr>
<td><strong>Plots</strong></td>
<td>mean 5.8 ± 0.6</td>
<td>3.7 ± 0.4</td>
<td>5.8 ± 0.6</td>
<td>3.8 ± 0.4</td>
</tr>
<tr>
<td></td>
<td>range 2 - 11</td>
<td>1 - 7</td>
<td>2 - 11</td>
<td>1 - 8</td>
</tr>
<tr>
<td></td>
<td>n 24</td>
<td>21</td>
<td>23</td>
<td>22</td>
</tr>
</tbody>
</table>
Table 6.5: Results from general ANOVA of 95% minimum convex polygons (MCP), span, captures and plots occupied by founder and wild-bred, and male and female eastern barred bandicoots caught at Hamilton Community Parklands between winter 2007 and autumn 2009. Span and MCP were \( \log_e \) transformed as the untransformed data were not normally distributed. Sex * Type is the interaction between the two.

<table>
<thead>
<tr>
<th>Response Variable</th>
<th>Factor</th>
<th>( F )</th>
<th>( P )</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \log_e ) MCP 95</td>
<td>Sex</td>
<td>16.4</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td>Type</td>
<td>6.3</td>
<td>0.016</td>
</tr>
<tr>
<td></td>
<td>Sex * Type</td>
<td>2.0</td>
<td>0.162</td>
</tr>
<tr>
<td>( \log_e ) Span</td>
<td>Sex</td>
<td>11.8</td>
<td>0.001</td>
</tr>
<tr>
<td></td>
<td>Type</td>
<td>9.3</td>
<td>0.004</td>
</tr>
<tr>
<td></td>
<td>Sex * Type</td>
<td>3.4</td>
<td>0.073</td>
</tr>
<tr>
<td>Captures</td>
<td>Sex</td>
<td>0.3</td>
<td>0.595</td>
</tr>
<tr>
<td></td>
<td>Type</td>
<td>7.3</td>
<td>0.010</td>
</tr>
<tr>
<td></td>
<td>Sex * Type</td>
<td>0.1</td>
<td>0.832</td>
</tr>
<tr>
<td>Plots</td>
<td>Sex</td>
<td>10.9</td>
<td>0.002</td>
</tr>
<tr>
<td></td>
<td>Type</td>
<td>15.0</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td></td>
<td>Sex * Type</td>
<td>0.9</td>
<td>0.362</td>
</tr>
</tbody>
</table>

Discussion

Understanding the habitat requirements of a critically endangered species is pivotal for successful recovery from extinction. In this study, 24 bandicoots were reintroduced into Hamilton Community Parklands in July 2007, and a further six bandicoots were added in November 2007. Foraging digs were found in both grassland and grassy woodland plots in each season and became spread across the entire reserve by winter 2008. Digs were more abundant in the grassy woodland plots in the reintroduction phase but this was only significant for winter 2007 (Fig 6.2), indicating that bandicoots initially stayed within the habitat type they were released in. In the established phase, digs became more prolific in the grasslands, but this did not differ significantly from the number of digs found in the woodlands (Fig 6.2).
Soil compaction was the main predictor of foraging digs in both the reintroduction and established phase, and this effect was strongest in winter 2008 (Table 6.1). Grass cover and distance to cover were only very weak predictors in the established phase and did not have a seasonal interaction with digs (Table 6.1). This strong relationship with soil compaction in winter 2008 could be due to the availability of soil invertebrates in this season. In cool, wet weather, soil invertebrates tend to be closer to the soil surface (Staley et al. 2007), whereas in the warmer months they move down the soil profile in response to high soil temperatures and low soil moisture (Reimer and Hindell 1996). Bandicoots not only eat soil invertebrates but a wide range of surface invertebrates, such as Coleoptera and Lepidoptera (see Heinsohn 1966, Reimer and Hindell 1996, Cook 2001). An increase in above-ground invertebrate prey, as soil invertebrates become less available (Reimer and Hindell 1996, Gibson 2001), could explain the weaker interaction between foraging digs and soil compaction in spring 2008, summer 2009 and autumn 2009. The lack of a seasonal interaction in the reintroduction phase is most likely due to the bandicoot population being present for only a short time.

In the reintroduction phase, male founder bandicoots had range spans more than three times larger than female founders (Table 6.2) and male founders travelled almost three times further from the release site than female founders. These greater movements of males made the whole of the Hamilton Community Parklands available to them, but not to female bandicoots, which stayed close to their release locations. The span and MCP of founder bandicoots in the reintroduction phase was not significantly different to their MCP and range span in the transition and established phases combined (Table 6.3). However, the non-significance of range span could be due to small sample size and consequent lack of statistical power.

In this study, the more extensive movements made by founders (males and females combined) compared with wild-bred bandicoots (Table 6.5) contradicted the prediction that newly released animals would limit their movements soon after release into this reserve. Furthermore, there was no significant difference between the movements made by founder bandicoots in the reintroduction phase compared to the post reintroduction phase (transition and established phases combined) (Table 6.2). This finding does not concur with Bright and Morris (1994), who found that reintroduced captive-bred dormice travelled shorter distances than resident dormice, and suggested that this was due to inexperience of natural habitats. Banks et al (2002) found the same pattern as Bright and
Morris (1994) in captive raised voles *Micrurus rossiaemeridionalis*. It is possible that restricted movements were not observed in this study as intensive monitoring was not achievable. The differences in ranging movements could also be an effect of small sample sizes, which is a common problem when studying a threatened species. However, males were found to have significantly larger MCP and range spans than females (Table 6.4) which is comparable with other studies of eastern barred bandicoots (Dufty 1991b, Dufty 1994b, Heinsohn 1966, Jenkins 1998, Ferguson 2006), suggesting that despite the low sample sizes, plausible patterns of spatial occupation have been obtained.

When first released, the broad spatial occupation of male founders could be explained by them attempting to increase their breeding opportunities, searching widely in response to the low density and patchy distribution of females, as suggested by Lynch et al. (2008) for male bobcats *Lynx rufus*. As the population density of bandicoots increased, due to an increase in the number of wild-bred individuals, the density of females would also increase as well as the number of aggressive interactions. Therefore wild-bred bandicoots could be occupying smaller areas due to a trade-off between more male-female mating interactions and more male-male aggressive interactions. This is suggestive of an ideal despotic distribution, where the value of each habitat is reduced by interference from dominant occupants (Morris 2003b). This kind of trade-off between a resource and aggression has been reported in Allenby’s gerbil (Ovadia and Dohna 2003). The fact that male and female founders did not decrease their area of occupation in response to increasing density could be due to them being dominant over younger wild-bred bandicoots. These explanations assume that males choose habitat based on access to females and no other resource such as food. As the eastern barred bandicoot is a generalist, opportunistic forager, it is conceivable that food is not a limiting factor for this species at Hamilton.

**Conclusion**

Habitats need to provide individuals with resources that best satisfy their requirements, including adequate food, water and suitable places to successfully reproduce and survive (Michel et al. 2008). Without these high quality habitats, translocation programs have a low chance of success, regardless of how many individuals are released or how well they are prepared for the release (Michel et al. 2008). The conditions at Hamilton Community Parklands at the time of release were suitable for eastern barred bandicoots as 28 of the 30 reintroduced bandicoots survived for up to 28 months and all females were known to
produce at least one litter. This study determined that when reintroduced into an empty, small reserve, males and females behave differently: males are capable of exploring the entire area whilst females restrict their movements to the area around the release location. This means that careful consideration should be given to the habitat in which animals are released. A difference in the spatial occupation of captive-bred and wild-bred bandicoots was also found, perhaps due to captive-bred bandicoots being dominant over the younger wild-bred bandicoots in an ideal despotic distribution.
Chapter 7

Habitat selection of a critically endangered species in a predator-free but degraded reserve

Overgrazed habitat at Mt Rothwell, September 2009
Chapter 7

Habitat selection of a critically endangered species in a predator-free but degraded reserve

Abstract

Reintroduction programs are often hampered by a lack of suitable release sites. In the case of the eastern barred bandicoot *Perameles gunnii*, a critically endangered grassland species in Victoria, Australia, successful releases are achieved only in fox-free areas. I studied a reintroduced population at Mt Rothwell, a 420-ha reserve surrounded by a predator-barrier fence. Since the first release of eastern barred bandicoots in 2004, habitat quality has declined significantly due to overgrazing by macropods. Despite this, the eastern barred bandicoot population appeared to be stable if not increasing. I radio-tracked 11 bandicoots for three weeks and conducted a habitat assessment at each nocturnal foraging and diurnal nesting location. The bandicoots were selecting the same habitat types for nesting and foraging as eastern barred bandicoots in fox-free habitat elsewhere, but home range size and distance moved in 24 h was the largest ever recorded (max 106.8 ha and 1.9 km respectively) for this species. Nest site use also differed from other sites; 13.2% of nests were used by at least two individuals, and bandicoots remained in the same nest for up to five nights, suggesting that there were limited nesting opportunities. This study highlights the behavioural flexibility of eastern barred bandicoots in response to structurally less complex habitat, and further demonstrates the importance of fox-free areas in reintroduction site selection.
Introduction

Habitat loss and degradation threatens more terrestrial species than any other process (Kingsford et al. 2009) and failure of reintroductions is inevitable if the habitat at the release site cannot support the species (Armstrong and Seddon 2008). Therefore it is imperative that high quality habitat be protected and maintained for successful reintroductions of threatened fauna (Griffith et al. 1989).

Habitat selection depends on the balance between rewards (e.g. food resources) and risk (e.g. predation) (Mayor et al. 2009). A decline in the availability of the most important resource can affect space use, in terms of increased movement rates and home range size, when suitable patches are further apart (Hansen et al. 2009, Pusenius and Viitala 1993, Mosnier et al. 2008). This can result in populations occurring at lower densities, and individuals having poorer body condition, smaller body size and participating less in breeding compared to their counterparts in habitats were resources are more plentiful and closely spaced (Smith et al. 2004, Mosnier et al. 2008). In the presence of predators, elevated vigilance reduces the time available for locating forage-rich sites and handling food, which has further consequences for energy budgets (Benhaiem et al. 2008). Poor body condition can also cause individuals to become more susceptible to risk, leading to mortality by predation, disease and exposure (Smith et al. 2004). In the absence of predators, foraging strategies are often based on maximising nutrient intake (Ripple and Beschta 2007); individuals can move through open areas more easily and consequently require smaller home ranges (Yunger 2004).

The eastern barred bandicoot *Perameles gunnii* is critically endangered in Victoria, Australia (Victorian DSE 2007). This small (<1 kg) insectivorous marsupial has declined to near extinction, apparently due to habitat loss and predation by introduced red foxes *Vulpes vulpes* (Winnard and Coulson 2008). This species is usually found in lowland native grassland, in which areas of high structural complexity are used for nesting, and open grasslands for foraging (Heinsohn 1966). The species was reintroduced to Mt Rothwell, a grassy woodland reserve surrounded by a predator-barrier fence, in 2004 (Winnard and Coulson 2008). This site is free of all introduced predators (red foxes and cats *Felis catus*) and at the time of the initial release, bandicoot foraging areas contained >74% grass cover (Ferguson 2006). Since then the reserve has been overgrazed, resulting in a substantial reduction in grass cover, which is quite evident although not quantified. As this reserve is
one of only three reintroduction sites where the eastern barred bandicoot occurs, understanding how they have adapted to this loss of cover is paramount to their recovery and has implications for choosing new release sites.

I investigated habitat use and spatial distribution of the eastern barred bandicoot at Mt Rothwell. There were two aims of this study: i) to quantify a set of habitat variables considered important for bandicoots, and ii) to determine if bandicoots exhibited any behavioural responses to the apparent habitat degradation. As habitat degradation was expected to reduce the availability of important resources, I expected home ranges to be larger than found elsewhere. Release from predation by exotic predators would allow eastern barred bandicoots to search areas more thoroughly for forage rich patches. As the main predators of eastern barred bandicoots at Mt Rothwell are diurnal raptors, I expected nesting to occur in the woodlands, which would provide some protection from these predators.

**Methods**

**Study Site**

Mt Rothwell is situated 60 km from Melbourne, Victoria, Australia, at the base of the You Yangs Ranges. It is a 420-ha reserve surrounded by a predator-barrier fence and contains a variety of mammals including the eastern barred bandicoot, southern brown bandicoot *Isoodon obesulus*, long-nosed potoroo *Potorous tridactylus*, rufous bettong *Aepyprymnus rufescens*, Tasmanian pademelon *Thyllogale billardierii*, eastern grey kangaroo *Macropus giganteus*, swamp wallaby *Wallabia bicolor*, common brushtail possum *Trichosurus vulpecula*, and eastern quoll *Dasyurus viverrinus*. The main potential predators are diurnal raptors such as wedge-tailed eagles *Aquila audax*. The reserve is divided into two zones, Zone 1 (170 ha) and Zone 3 (250 ha), by a 400-m internal fence (Fig 7.1). Bandicoots can move between the two zones via 30 PVC tubes, which had been placed in the base of the internal fence, 10 m apart, six months prior to this study. The habitat in Zone 1 is classified as grassy woodlands and has been heavily grazed by overabundant populations of eastern grey kangaroos and Tasmanian pademelons, resulting in very little grass cover. The most common tree species in Zone 1 are yellow box *Eucalyptus melliodora* and red box *E. polyanthemos*, with hedge wattle *Acacia paradoxa*, dominating the shrub layer and spear
grasses *Stipa* spp. the most common ground cover. Zone 3 is an open grassland habitat with basaltic stony rises. Grazing pressure is less in this zone but the habitat is not considered optimal for eastern barred bandicoots due to low grass cover. This zone is dominated by spear grasses, and there is a patchy distribution of black wattle *A. mearnsii*, Golden wattle *A. pycnantha* and hedge wattle *A. pycnantha*

![Fig 7.1: Mt Rothwell showing the areas of woodland (darker areas) and grassland (paler areas). Zone 3 includes an area known as the Rocky Rises. The yellow line indicates the predator-barrier fence, whilst the pale line between Zone 1 and 3 indicates the internal fence which allows access by small mammals into each zone. Aerial photograph supplied by the Department of Sustainability and Environment.](image)

**Trapping**

In autumn 2009, I trapped for 14 days over a three week period, using 50 x 18 x 20-cm (length x width x height) wire cage traps baited with peanut butter, honey and rolled oats. I placed 75 traps on two grids covering ~1 ha each in areas where I had observed an eastern barred bandicoot whilst spotlighting. On each grid I placed traps approximately 7 m apart. I changed the location of both grids daily and checked traps at night between 2200 and 2400
and again in the morning from 0700. Once an eastern barred bandicoot was caught I scanned it for a Trovan PIT (passive integrated transponder) tag, weighed it, checked its body condition, looked for any parasites and injuries and then attached a 4-g radio transmitter to its tail with Fixomull stretch tape. Transmitters were single-stage tail mount transmitters with a pulse rate of 40 min\(^{-1}\) (Sirtrack Ltd, New Zealand), which took ≤2 min to attach with Fixomull stretch tape. This technique was chosen over other techniques (e.g. collars) due to difficulties encountered in previous studies with short attachment periods and a high level of injuries (Murphy and Serena 1993).

In total I trapped nine adult eastern barred bandicoots (2 female and 7 male) in 1,050 trap nights. This low capture rate was due to a very high by-capture rate (48.4%, \(n = 508\) non target species), particularly of rufous bettongs, and trap interference from these other species. I caught another two males by hand when I accidently stood on their cryptic nests.

**Radio-tracking**

From the time of capture, I tracked each bandicoot once during its active (night) and once during its inactive (day) period every 24 h, using a Communications Specialists Inc. R-1000 telemetry receiver and Yagi AY/C antenna. At night I used a 50-W spotlight to locate tagged bandicoots. In the majority of cases I could approach within 1 - 10 m of a bandicoot before it fled. I then marked the location with a tent peg and recorded it using a handheld GPS unit (Garmin GPS60). I tracked bandicoots at different times every night, so obtained an approximately even distribution of locations throughout the night. During the day, I marked nest sites with a tent peg, which I placed approximately 20-cm from the nest so I did not disturb the bandicoot; I then recorded the location as before. Radio-transmitters remained attached for 7 - 33 days, resulting in a range of 5 - 47 locations for males and 12 - 25 locations for females. In total I obtained 140 nest locations, 102 foraging locations and 12 trap locations.

**Habitat Analysis**

At each foraging (night) and nesting (day) location I conducted a habitat assessment. I assessed habitat at nest sites only when the individual that had occupied that nest was known to be nesting elsewhere. I sampled circular plots with a radius of 10 m centred on each nesting and foraging location. Within each plot I made a visual estimate of the percentage (to the nearest 5%) cover of grass in two height categories (<15 cm and >15 cm;
the approximate height of an eastern barred bandicoot), woody debris (twigs and sticks), litter (leaf and grass), moss, rock and bare ground. I then counted the number of trees within the plot and the area covered by coarse woody debris (CWD; one or more pieces of woody material where at least one piece was ≥0.5 m long) which was then categorized into one of five cover categories: category 1 reflected no cover (e.g. small branch or pile of twigs) and category 5 the most cover (e.g. fallen tree or large limbs with many branches and cavities). I also measured the distance to the nearest cover (defined as anything that could conceal an eastern barred bandicoot, such as a tree, CWD or hedge wattle) and took five soil compaction readings, using a pocket penetrometer (Geotest Instrument Corp), at the centre of the plot and at the four cardinal directions 10 m from the centre. Using a Fujifilm FinePix S5500 digital camera, I took photographs of a 25 x 15-cm (approximate dimensions of an eastern barred bandicoot) red cover board at each cardinal direction, 10 m from the centre of the plot, and a fifth photograph from the centre of the plot looking vertically to determine the projective foliage cover (PFC). I later analysed these images in Adobe Photoshop CS, by counting the number of vegetation and cover board pixels (or sky pixels for PFC), then calculating the percentage lateral cover and PFC. I also collected soil samples to determine the percentage soil moisture, but did not include this variable in the analysis due to little variance in soil moisture across the reserve. This was also confounded by rainfall occurring in the interval between obtaining locations and habitat sampling.

I did not quantify habitat at trap sites as the use of these areas was influenced by the presence of bait. Four foraging locations were also removed from all analyses as they were in a 12-ha area containing 3-year-old acacias and eucalypts and a mix of native shrubs, and this habitat could not be classified as grasslands or woodlands.

I calculated the proportional habitat use of each bandicoot by summing the number of grassland and woodland locations and dividing by the total number of locations. I then used ArcGIS 9.3 to calculate the proportion of habitat available to bandicoots at two scales: study area and home range. I determined the proportion of habitat type (grassland and woodland) available using an aerial photo of Mt Rothwell, then drew the smallest circle that encompassed all locations, which I layered onto the aerial photo. I set the area it encompassed as the total area available to all bandicoots in this study. This area encompassed all of Zone 1 and an area known as the Rocky Rises in Zone 3 (Fig 7.1), which was the only part of Zone 3 that tagged bandicoots used. I then determined the proportion of grassland and woodland within this circle from the aerial photograph. If the circle
crossed the boundary fence, I disregarded the area outside the fence, as it was not accessible by eastern barred bandicoots. I employed the same method to determine the proportion of habitat available at the home range scale, and drew circles around all locations for each individual eastern barred bandicoot. I then calculated the proportion of grasslands and woodlands within each circle.

**Data Analysis**

I used the selection index $B$ (Manly et al. 2002) to compare the use and availability of nesting and foraging sites at both the study area and home range scale. I calculated an index value separately for each individual in each habitat type (grassland and woodland), and used a one-way ANOVA, using each individual bandicoot as a random factor in the analysis, to test whether the mean selection indices differed for foraging or nesting in each habitat type, using GenStat 12.

I graphically represented habitat variables from locations used by bandicoots using ordination (non-metric multidimensional scaling) in Primer 6. I created a Euclidean resemblance matrix on normalised data, and some variables were log$_{10}$ or square root transformed. Transformations were used to reduce the influence of outlying variables and for consistency with the generalised linear mixed model (GLMM; see below). The output reflects differences between foraging and nesting locations. To test whether this difference was significant, I ran an ANOSIM in Primer 6.

I used logistic regression in GenStat 12 to determine the effect of habitat variables (% lateral cover, >15 cm grass, <15 cm grass, litter, woody debris, CWD (categories 3 – 5), PFC and distance to cover) on bandicoot activity (foraging or nesting). I used a GLMM to take the nested nature of the data set (locations nested within bandicoots) into account, treating individual bandicoots as a random factor in the model. A number of variables were skewed to the right (i.e. not normally distributed), so I either log$_{10}$ (>15 cm grass, woody debris) or square root (litter) transformed them to improve the symmetry of the distributions. Other variables contained a substantial number of zero values, so I converted these into categorical variables (CWD, PFC, distance to cover). I calculated Pearson’s correlation coefficient between all the continuous variables and excluded two variables from the final analysis due to substantial correlations: grass <15 cm, which was correlated with log$_{10}$ woody debris ($r = -0.72$) and bare ground which was correlated with square root litter ($r = -0.71$). The correlation coefficients for all other variable combinations were
between -0.51 and +0.64. The final set of variables I considered was % lateral cover, \( \log_{10} >15 \) cm grass, \( \log_{10} \) soil compaction, square root litter, \( \log_{10} \) woody debris, CWD (presence/absence), PFC (presence/absence), distance to cover (zero, not zero).

I developed models manually using a backwards elimination approach, constructing the most parsimonious model by deleting terms that did not contribute significantly to the explained variance at a probability level of 5%. I deleted PFC and soil compaction from the model as they did not contribute at the 5% level. I also deleted CWD, which was a significant predictor in the model but this effect seemed to be an artifact of the way it combined with the other variables, based on two lines of evidence. First, if CWD was entered into the model by itself, it was not a good predictor \((P = 0.53)\), unlike all other variables included in the final model. Second, the CWD raw data did not discriminate between foraging and nesting locations. I also deleted woody debris from the model as it had a reasonably strong correlation with litter \((r = +0.64)\), which was causing the direction of the effect to reverse. The final model reflecting the contrast between foraging and nesting locations included % lateral cover, \( \log_{10} >15 \) cm grass, square root litter and distance to cover (zero, not zero). I identified a few points of high leverage in the model, but re-running the analysis after removing them had little influence on the result.

**Nest use**

I calculated duration of nest use by dividing the total number of days I tracked each bandicoot by the total number of nests that individual occupied. I then plotted the total number of nests known to be used by each bandicoot with the number of days radiotracked. I calculated the percentage of nests reused over the study period for each bandicoot, only including nests that were re-occupied after a bandicoot had moved to another nest. I used Ranges7 to calculate distance between consecutive nests, including only the nests located on consecutive days in the analysis.

**Home range**

I used Ranges7 to calculate home range using minimum convex polygons (MCP) with 95% isopleths and harmonic means, and kernel contours (KL) with 95% isopleths and a reference bandwidth \((h_{\text{Ref}})\) multiplied by a 0.75 smoothing multiplier for comparison with Ferguson (2006). I chose \(h_{\text{Ref}}\) over least-squares cross-validation as the latter technique greatly over estimates home range area when fewer than 50 locations are used (Seaman et al. 1999).
used MCP so that these data could be compared to previous studies, and KL because it is currently the more widely accepted method for calculating home range. I also calculated maximum distance travelled in 24 h using Ranges7. This measurement is the distance a bandicoot travelled between nest site and foraging site in any 24-h period. Finally I calculated range span, the greatest distance between locations of an individual, using Ranges7.

Results

Habitat Use

At all locations where bandicoots were found, most grass cover was low to the ground and ranged from 0 - 63% cover, with a mean ± SE of 23 ± 1%. Bare ground was common at all locations and ranged from 15 to 90% cover with a mean of 38 ± 1%. Despite the lack of grass cover at Mt Rothwell, bandicoots were utilising areas with plenty of lateral cover, ranging from 6 - 99% with a mean of 55 ± 2% for foraging and nesting sites combined. Grassland was favoured for foraging and woodland for nesting at both the home range (foraging $P = 0.020$, nesting $P = 0.005$) and study area (foraging $P = 0.045$, nesting $P = 0.007$) scale; these habitat types were strongly selected for irrespective of availability (Fig 7.2). An example of the separation of forage and nest sites, in relation to home range, seen in most bandicoots in this study, is given in Fig 7.3.

Table 7.1 shows the difference in occurrence of nine habitat variables at foraging and nesting sites. Foraging sites had a higher percentage of grass cover in both height categories (<15 and >15 cm), were further from cover and had more compact soil than nest sites (Table 7.1). Nest sites had a higher percentage of lateral cover, litter, woody debris, PFC and contained more CWD than foraging sites (Table 7.1). Ordination of the habitat variables in two dimensions showed a difference in habitat variables selected for foraging and nesting (Fig 7.4), which was significant when run in an ANOSIM ($r = +0.35$, $P = 0.001$).
Fig 7.2: Proportional habitat use at the home range and study area scale for foraging and nesting at Mt Rothwell in spring 2009. $B$ is the probability that a habitat (woodland or grassland) will be used next for a particular activity (foraging or nesting) assuming that both habitat types were equally available. A value $>0$ on the $y$-axis means that woodland was selected over grassland. A value $<0$ on the $y$-axis means that grassland was selected over woodland. Error bars are 95% confidence intervals. Error bars that do not cross zero indicate that $B$-values are likely to be greater or smaller than zero and these were all significant ($P < 0.05$).

Fig 7.3: Kernel contours home range of a typical male (PIT tag # 0006C9754F) in this study, showing the location of foraging and nesting sites at Mt Rothwell in spring 2009.
Table 7.1: Means of foraging and nesting habitat variables at Mt Rothwell in autumn 2009 ± standard error

<table>
<thead>
<tr>
<th>Habitat variable</th>
<th>Foraging</th>
<th>Nesting</th>
</tr>
</thead>
<tbody>
<tr>
<td>% lateral cover</td>
<td>41.6 ± 2.4</td>
<td>64.6 ± 1.5</td>
</tr>
<tr>
<td>% &lt;15 cm grass</td>
<td>25.4 ± 1.3</td>
<td>14.3 ± 1.2</td>
</tr>
<tr>
<td>% &gt;15 cm grass</td>
<td>5.5 ± 0.5</td>
<td>3.1 ± 0.3</td>
</tr>
<tr>
<td>% litter</td>
<td>10.6 ± 1.2</td>
<td>27.5 ± 1.2</td>
</tr>
<tr>
<td>% woody debris</td>
<td>10.3 ± 1.2</td>
<td>19.3 ± 0.9</td>
</tr>
<tr>
<td>% projective foliage cover</td>
<td>7.8 ± 1.9</td>
<td>28.7 ± 2.7</td>
</tr>
<tr>
<td>Distance to cover (m)</td>
<td>15.3 ± 1.6</td>
<td>5.4 ± 1.3</td>
</tr>
<tr>
<td>Soil compaction (kg/cm²)</td>
<td>1.15 ± 0.07</td>
<td>0.96 ± 0.05</td>
</tr>
<tr>
<td>% plots with CWD*</td>
<td>11</td>
<td>8</td>
</tr>
</tbody>
</table>

*Coarse woody debris (CWD) was transformed into a categorical variable (presence/absence) therefore this variable is presented as the percentage of foraging and nesting plots that contain CWD.

Fig 7.4: Ordination diagram created using non-metric multidimensional scaling and a Euclidean resemblance matrix. Points on the diagram represent foraging and nesting locations at Mt Rothwell in autumn 2009. Variables used were % lateral cover, <15 cm grass, square root litter, $\log_{10} >15$ cm grass, $\log_{10}$ soil compaction, $\log_{10}$ woody debris, $\log_{10}$ distance to cover and $\log_{10}$ projective foliage cover. Some variables were transformed to reduce the influence of outlying variables and for consistency with the generalised linear mixed model (or because presence/absence data could not be used). Data were normalised prior to analysis.
The final model quantifying the influence of habitat variables on foraging and nesting habitat selection by the sampled bandicoots is summarised in Table 7.2. Foraging site selection was positively correlated with >15 cm grass. Nest site selection was positively correlated with % lateral cover, % litter and locations where distance to cover equalled zero. The odds ratio in Table 7.2 reflects the strength of these correlations. For example, the odds of a bandicoot occupying a foraging rather than a nesting location decreases by a factor of 0.54 (or 46%) for every 1% increase in square root litter, provided that all other variables remain constant. Although there was a statistically significant effect of \log_{10} >15 \text{ cm grass} it did not discriminate strongly between foraging and nesting locations, and there was a degree of uncertainty associated with the estimate (Table 7.2).

<table>
<thead>
<tr>
<th>Variable</th>
<th>Estimate</th>
<th>SE</th>
<th>OR</th>
<th>LCL</th>
<th>UCL</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sq root litter</td>
<td>-0.62</td>
<td>0.14</td>
<td>0.54</td>
<td>0.41</td>
<td>0.70</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>\log_{10} &gt;15 \text{ cm grass}</td>
<td>2.15</td>
<td>0.85</td>
<td>8.55</td>
<td>1.60</td>
<td>45.61</td>
<td>0.012</td>
</tr>
<tr>
<td>Lateral cover</td>
<td>-0.03</td>
<td>0.01</td>
<td>0.97</td>
<td>0.95</td>
<td>0.99</td>
<td>0.001</td>
</tr>
<tr>
<td>Distance to cover</td>
<td>-2.38</td>
<td>0.65</td>
<td>0.09</td>
<td>0.03</td>
<td>0.33</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

Nest use

I identified four different nest types in this study. The most common type (53%) was constructed from leaf litter, was level with the ground, highly cryptic and found in woodland areas, usually next to a tree or CWD. The other nest type (11%) constructed in the woodlands, next to CWD, was a large mound made from leaf litter and soil; this was also cryptic but easily found when radio-tracking. In the open grasslands, 30% of nests were
constructed by digging a shallow depression in the ground, filling it in with grass and piling a substantial amount of grass on top, although 6% of nests were covered with soil instead of grass. All grassland nests were level with the ground and cryptic, yet more easily located than leaf litter nests due to the lack of grass at Mt Rothwell. Only 4% of nests were constructed under hedge wattle, and another 2% were under a pile of planks. Both nest types were located in the open grasslands, but it is not known what they were constructed of, as I could not find their exact location.

Of the 53 nests located in this study, 49% were used on more than one occasion. Some nests (13%) were known to have at least two different occupants at different times; one nest had three known occupants. Overall, 64% of nests were located in woodlands and the remainder in grasslands. Woodland nests were used an average ± SE of 3.2 ± 0.4 days, whereas grassland nests were used for only 1.4 ± 0.2 days ($t = 3.81$, $df = 42.7$, $P < 0.001$). Males tended to spend more days in a nest and reuse more nests than females, but these differences were not significant (Table 7.3). The number of nests occupied by each bandicoot tended to increase linearly with time, but this relationship was weak and non significant ($r^2 = +0.22$, $P = 0.145$) (Fig 7.5).

<table>
<thead>
<tr>
<th></th>
<th>Duration of nest use (days)</th>
<th>% nest reuse</th>
<th>Distance between consecutive nests (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Male</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>mean</td>
<td>2.8 ± 0.4</td>
<td>17.9 ± 7.1</td>
<td>122.6 ± 29.0</td>
</tr>
<tr>
<td>range</td>
<td>1 – 4.3</td>
<td>0 – 50</td>
<td>0 – 1765.7</td>
</tr>
<tr>
<td>$n$</td>
<td>9</td>
<td>9</td>
<td>89</td>
</tr>
<tr>
<td><strong>Female</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>mean</td>
<td>1.8 ± 0.3</td>
<td>14.3 ± 14.3</td>
<td>125.2 ± 30.3</td>
</tr>
<tr>
<td>range</td>
<td>1.5 – 2</td>
<td>0 – 28.6</td>
<td>0 – 403.1</td>
</tr>
<tr>
<td>$n$</td>
<td>2</td>
<td>2</td>
<td>15</td>
</tr>
</tbody>
</table>

Table 7.3: Summary of nest use by male and female eastern barred bandicoots at Mt Rothwell in autumn 2009. Means are ± standard error.
Fig 7.5: Linear regression of the total number of nests known to be used by each eastern barred bandicoot \((n = 11)\) at Mt Rothwell in autumn 2009 on the number of days radio-tracked; \(r^2 = 0.22, P = 0.145\).

**Home range**

Mean male home range (MCP and KL) was more than twice the size of mean female home range, but these were not significant differences (Table 7.4). Males also tended to travel further than females from their nest to forage site, and have range spans almost twice those of females, but these differences were also not significant (Table 7.4).
Table 7.4: Home range, range span and maximum distance travelled in 24 hours by male and female eastern barred bandicoots at Mt Rothwell in autumn 2009. Means are ± standard error. The t-tests examine the differences between males and females within each variable.

For male minimum convex polygons (MCP) and kernel contours (KL) home range and range span, one eastern barred bandicoot was omitted from calculations due to having a low number of locations (n = 5). His data were only used in the calculation of maximum distance travelled in 24 h.

<table>
<thead>
<tr>
<th></th>
<th>Home range 95% MCP (ha)</th>
<th>Home range 95% KL (ha)</th>
<th>Distance b/w nest &amp; forage site (m)</th>
<th>Range span (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Male</strong></td>
<td>mean 24.2 ± 9.6</td>
<td>37.2 ± 11.8</td>
<td>494.6 ± 32.5</td>
<td>1045.2 ± 196.7</td>
</tr>
<tr>
<td></td>
<td>range 1.6 – 85.5</td>
<td>3.5 – 106.8</td>
<td>49.3 – 1851.8</td>
<td>210.9 – 2117.6</td>
</tr>
<tr>
<td></td>
<td>n 8</td>
<td>8</td>
<td>185</td>
<td>8</td>
</tr>
<tr>
<td><strong>Female</strong></td>
<td>mean 8.9 ± 7.0</td>
<td>15.8 ± 13.0</td>
<td>330.0 ± 44.0</td>
<td>620.0 ± 215.1</td>
</tr>
<tr>
<td></td>
<td>range 1.7 – 15.7</td>
<td>2.8 – 28.8</td>
<td>24.8 – 794.0</td>
<td>404.9 – 835.1</td>
</tr>
<tr>
<td></td>
<td>n 2</td>
<td>2</td>
<td>32</td>
<td>2</td>
</tr>
<tr>
<td><strong>t-test</strong></td>
<td>t 0.77</td>
<td>0.85</td>
<td>0.76</td>
<td>1.01</td>
</tr>
<tr>
<td></td>
<td>df 8</td>
<td>8</td>
<td>8</td>
<td>8</td>
</tr>
<tr>
<td></td>
<td>P (2 tailed) 0.46</td>
<td>0.42</td>
<td>0.47</td>
<td>0.34</td>
</tr>
</tbody>
</table>

**Discussion**

Mt Rothwell is a 420-ha grassy woodland reserve surrounded by a predator-barrier fence. In recent years, the reserve has suffered from over grazing by macropods causing a decline in the amount of grass cover. This study reported a mean grass cover of 31% in eastern barred bandicoot foraging patches, compared to >74% cover recorded by Ferguson (2006) at Mt Rothwell in 2005. Dufty (1991b) found that bandicoots selected habitats with high ground cover and tall grass height for foraging. Heterogeneous and structurally complex areas that possess high floristic diversity due to a lack of grazing are also considered suitable habitat for eastern bared bandicoots (Dufty 1994b). Despite Mt Rothwell becoming a structurally less complex habitat overall, the bandicoots in this study were nesting in the more structurally complex areas and foraging in open areas, which is consistent with my

Dense ground cover for nest sites and shelter from predators is considered to influence the suitability of habitat for eastern barred bandicoots (Mallick et al. 1997a, Heinsohn 1966, Dufty 1991b). At Mt Rothwell there was very little ground cover available, but trees, shrubs and CWD did provide cover for nests. Foraging locations were associated with open areas, far from trees, shrubs or CWD. Distance between consecutive nests was larger in this study than reported elsewhere; Jenkins (1998) 25 ± 28 m (± SD); Ferguson (2006) 68.4 ± 15.4 m (± SE) and 108.7 ± 10.2 m. Previous studies have also found that nests are not confined to core areas of the home range (Jenkins 1998, Dufty 1991b); bandicoots tend to nest in or near tree or shrub cover, but nesting in open areas is not uncommon (Dufty 1991b, Ferguson 2006, Heinsohn 1966, Jenkins 1998, Murphy and Serena 1993). Nest type has been found to vary depending on where they are constructed, with eastern barred bandicoots compensating for a lack of structural complexity by making their nests more cryptic (Dufty 1991b, Murphy and Serena 1993, Heinsohn 1966). As in other studies, eastern barred bandicoots in this study nested throughout their home range but usually constrained nesting to woodland areas. Nesting in the open grasslands was uncommon in this study and most likely due to a lack of litter and other vegetation, such as grass tussocks, which are required for nest construction. The majority of nests were also highly cryptic in this study, probably to compensate for the lack of ground layer vegetation.

On average, eastern barred bandicoots have been found to change nest site every two days (Ferguson 2006, Jenkins 1998). Prolonged nest use was reported by Dufty (1991b), who recorded one female eastern barred bandicoot in the same nest for six consecutive nights. In this study, 30% of nests were occupied for more than two consecutive days and the maximum length of known nest occupation was five days. However, as radio tracking did not occur on every day there is a high probability that the real figures were higher. For example, one bandicoot occupied the same nest on seven occasions in eight days; his nest was not located on day 4. Therefore it is unknown if he switched nest on day 4 and then returned. The number of nests a bandicoot used over time showed a weak, positive relationship that was not significant (Fig 7.5). Nest disturbance has been suggested as a reason for changing nest location as disturbance is likely to be caused by predators (Jenkins 1998, Ferguson 2006, Heinsohn 1966). Therefore, a lack of nest disturbance could lead to prolonged nest occupation.
Re-use of nests on consecutive days is commonly reported in eastern barred bandicoots (Murphy and Serena 1993, Ferguson 2006). I also found that different eastern barred bandicoots would occupy the same nest on different days. This has never been reported before for eastern barred bandicoots but has been observed in another reintroduced Peramelid, the greater bilby *Macrotis lagotis* (Moseby and O'Donnell 2003). Active defence of nests by eastern barred bandicoots was observed by Heinsohn (1966) and Jenkins (1998), but my findings suggested that nests were not defended in this population. This was possibly due to a lower density of suitable nest sites, which were spaced further apart and would be difficult to maintain exclusive use of, due to increased travel times. A lack of suitable nests could also explain the prolonged nest occupation in this population, and occasional multiple occupants in some nests.

Mean male home range size in this study was larger than that found by Ferguson (2006), Jenkins (1998) and Dufty (1994b) but comparable to Heinsohn (1966), whilst mean female home range is larger than reported in these studies (Table 7.5). Home ranges of three males and one female in this study were larger than the maximum previously recorded for eastern barred bandicoots. Maximum distance travelled in 24 h in this study was also larger than previously reported for both sexes (632.8 m and 472.7 m for males and females respectively: Heinsohn 1966). Larger distances have been recorded, but these were reported as dispersal movements that occurred over several weeks (Dufty 1991b). In my study, one bandicoot was travelling in excess of 1.5 km per night between foraging and nest sites.

The size of home ranges and distances travelled in 24 h can be interpreted as a response to the less complex habitat at Mt Rothwell, high quality forage patches and suitable nest sites being difficult to find in close proximity to each other. In environments where food resources are limited, habitat selection usually reflects a quest for the highest possible net energy gain (Mosnier *et al.* 2008). Although I did not examine food availability, it has been found that in predator free areas, prey species will maximise their energy intake by foraging in the highest quality patches, which they can search more efficiently, resulting in smaller home ranges (Yunger 2004). This pattern is seen in the southern brown bandicoot, where home range size is negatively correlated to invertebrate food abundance (Broughton and Dickman 1991). Southern brown bandicoots have the capacity to detect and respond rapidly to rich sources of food outside their home range, and may expand their ranges to exploit them (Broughton and Dickman 1991). Eastern barred bandicoots may also have this
ability, as suggested by the seven bandicoots that foraged in the recently accessible Rocky Rises of Zone 3.

Table 7.5: Home ranges calculated in previous studies for male and female eastern barred bandicoots using minimum convex polygons (MCP) or kernel contours (KL).

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Location</strong></td>
<td>Tasmania</td>
<td>Hamilton</td>
<td>Woodlands Historic Park</td>
<td>Mt Rothwell</td>
</tr>
<tr>
<td><strong>Method</strong></td>
<td>Trapping</td>
<td>Trapping</td>
<td>Radio-tracking</td>
<td>Radio-tracking</td>
</tr>
<tr>
<td><strong>Male</strong></td>
<td>mean 26.3 ± 4.6</td>
<td>4.0 ± 0.6</td>
<td>4.9 ± 1.5</td>
<td>21.0 ± 7.4</td>
</tr>
<tr>
<td></td>
<td>range 18.7 – 39.7</td>
<td>0.8 – 9.0</td>
<td>1.9 – 6.4</td>
<td>6.2 – 32.5</td>
</tr>
<tr>
<td></td>
<td>n 4</td>
<td>18</td>
<td>12</td>
<td>4</td>
</tr>
<tr>
<td><strong>Female</strong></td>
<td>mean 3.2 ± 1.1</td>
<td>1.6 ± 0.4</td>
<td>n/a</td>
<td>8.1 ± 2.2</td>
</tr>
<tr>
<td></td>
<td>range 0.9 – 11.4</td>
<td>0.1 – 5.9</td>
<td></td>
<td>0.1 – 20.2</td>
</tr>
<tr>
<td></td>
<td>n 9</td>
<td>13</td>
<td></td>
<td>8</td>
</tr>
</tbody>
</table>

**Conclusion**

Although Mt Rothwell is lacking in ground vegetation, it does provide enough structure for eastern barred bandicoots and my study shows that this species is behaviourally flexible. Spatial use of habitat by the eastern barred bandicoots in this study did not differ from other studies conducted in better quality, fox-free areas; only the scale of habitat use differed. The ability to cope with habitat alteration has also been seen in another Peramelid, the greater bilby (Moseby and O'Donnell 2003). The success of bilby reintroductions were suggested to be more dependent on the strict control or removal of introduced predators rather than the quality of the habitat (Moseby and O'Donnell 2003).
This study has implications for future eastern barred bandicoot reintroduction site selection, because habitats that have previously been considered poor quality and therefore unsuitable could support a population of eastern barred bandicoots, provided that it remains fox free. Despite this finding, it is important to prevent current reintroduction sites from becoming degraded and to improve the habitat quality at new sites, as there is potential for reserves to become further degraded by food depletion, which would negatively affect bandicoot populations.
Chapter 8

Management

*Eastern barred bandicoot road sign in Hamilton, January 2009*
Chapter 8

Management

Future management for the eastern barred bandicoot

Many species worldwide are at risk of extinction. Diamond (1989) classed the agents of population decline into four groups: overkill (from hunting at a rate above the maximum sustained yield), habitat destruction and fragmentation, impact of introduced species and chains of extinction (when the extinction of one species is caused by the extinction of another upon which it depends). In the case of the eastern barred bandicoot *Perameles gunnii*, predation by introduced red foxes *Vulpes vulpes* and cats *Felis catus*, combined with habitat degradation and loss, are thought to be the main agents that caused the demise of this species (Backhouse *et al.* 1994).

Reintroduction projects aim to re-establish viable, free-ranging populations of endangered species within their historical range, through the release of wild or captive-bred individuals following extirpation or extinction in the wild (IUCN 1998). These projects can be costly and time consuming, and are often not successful (Benson and Chamberlain 2007). Therefore, understanding the factors that contribute to the success or failure of reintroductions is essential to progress reintroduction biology as a conservation tool (Banks *et al.* 2002). According to Teixeira *et al.* (2007) translocation and reintroduction of animals have three main objectives: *i*) survival of the animals after release; *ii*) settlement of the animals into the release area and *iii*) successful reproduction in the release area. It is also essential that the factors that caused the original species’ decline are controlled or eliminated before the reintroduction is attempted (Kleiman 1989). These factors could include, improving the general habitat quality, climatic conditions or removing predators or competitors (Wolf *et al.* 1998).

In this PhD I aimed to gain the knowledge needed to improve the success of the eastern barred bandicoot reintroduction program. Based on my findings I am able to recommend a range of management actions that are organised into four sections: predators, habitat, releases and monitoring.
Introduced Predators

Control of introduced predators is often the most essential management technique for the successful re-establishment of threatened species (Moseby et al. 2009). Whilst I did not examine the relationship between introduced predators and eastern barred bandicoots directly, the importance of releasing eastern barred bandicoots into predator-free habitats was highlighted throughout this thesis. Firstly, Chapter 2 documents the eastern barred bandicoot recovery program from when it first began up to 2005. Of the eight sites in which eastern barred bandicoots were released, six underwent population declines and extinctions. Whilst predation by introduced red foxes was not thought to be entirely to blame at each site, it is the only known factor that these sites have in common. This is not unusual, as mortality caused by predation is known to be a significant factor limiting the survival of many released animals (Seddon et al. 2007). Therefore, I agree with Short et al. (1992) that control of predation pressure should be considered an essential component of reintroduction programs.

Mooramong is an anomaly in the history of bandicoot reintroductions into areas containing red foxes. Bandicoots have persisted at this site for at least 17 years with no ‘top up’ releases. Despite regular and ongoing fox control, Chapter 4 concluded that this population is mostly regulated by mortality, for which the most likely source is red foxes, although no direct evidence of this was found. This predation pressure probably caused the population to contract to the confines of the Homestead Gardens, which were presumably too small or unsuitable to support a self-sustaining population, so a progressive decline of this population ensued. This scenario is in agreement with Caughley’s (1994) statement that a species will end up, not in the habitat most favourable to it, but in the habitat least favourable to the agent of decline.

Whilst I consider that the long-term goal of the eastern barred bandicoot recovery program should be multiple self-sustaining wild populations that are not kept within the confines of a protective predator-barrier fence, I believe that we are a long way from achieving that goal. My findings support the current reintroduction plan (Hill et al. 2010) and I recommend, that no eastern barred bandicoots are released into areas containing foxes in the short term, even if an intensive fox control program is in place. This species must be secure and numbers increased dramatically in predator free areas before consideration is
given to releasing bandicoots into areas containing foxes. A strict monitoring regime with clear management objectives should then ensue for all released individuals.

**Habitat**

Worldwide, the decline of suitable habitat has been a major factor contributing to the decline and extinction of many species (Michel et al. 2008). Wolf et al. (1998) emphasized that habitat quality was as an essential component of translocation success. My findings do not concur with their conclusion. Chapter 7 provides evidence that habitat quality is not a necessary requirement for a successful eastern barred bandicoot reintroduction into fox-free habitat. This finding, and the ability of the species to cope with habitat alteration, is in agreement with Moseby and O’Donnell (2003), who concluded that the success of greater bilby *Macrotis lagotis* reintroductions were more dependent on strict control and removal of introduced predators than on the quality of the habitat.

My research at Hamilton Community Parklands (Chapter 6) was in habitat which, from a human perspective, looked more favourable than Mt Rothwell, due to the grass cover being tall and dense throughout the reserve. The majority of released captive-bred bandicoots (93%) survived the initial reintroduction at Hamilton, as was evident from the trapping record (Chapter 6). This high survival rate may be due to the seemingly higher quality of the habitat, although the measured variable that contributed most to foraging dig abundance was soil compaction, suggesting that eastern barred bandicoots are not highly selective of vegetation structure when foraging.

My findings have important implications for selection of future reintroduction sites. As eastern barred bandicoots can adapt and survive in habitats that appear to be sub optimal, emphasis on future reintroduction sites should be on maintaining areas that are free of introduced predators, particularly red foxes. If this can be achieved and maintained long term, then any habitat rehabilitation can occur after a reintroduction has taken place. The basic requirements of eastern barred bandicoots are structurally complex areas, such as woodland for nesting and open areas for foraging. Therefore, as long as these criteria are met, releases do not need to be postponed until the habitat quality is perceived to be of a high quality. However, this is not an argument for not rehabilitating habitat within eastern barred bandicoot reserves. This should continue to be strived for, as there may be a relationship between habitat quality and carrying capacity of the reserve.
Releases

Wolf et al. (1998) considered a relatively high reproductive potential, an omnivorous diet, small body mass, and high genetic diversity to be favourable characteristics for successful invasion of a new location by released individuals. Whilst the latter of these characteristics is probably not applicable for the eastern barred bandicoot (Weeks 2010), the other three are. To take advantage of these characteristics, and to improve the success of bandicoot reintroductions, the release location should be carefully chosen. As animals unfamiliar with a reintroduction site typically move very little soon after release (Banks et al. 2002), the habitat they are released into needs to provide individuals with resources that best satisfy their requirements, such as food, water and suitable places to successfully reproduce and survive (Michel et al. 2008).

Based on the results of Chapter 6, I recommend that eastern barred bandicoots, particularly females, which did not move as much as males, are released into woodland habitats close to the woodland-grassland ecotone. This will enable them to readily exploit woodland areas for nesting and open grassland areas for foraging, as occurred at Mt Rothwell (Chapter 7). A release into woodland areas would also provide some protection from aerial predators such as wedge-tailed eagles Aquila audax.

The time of year in which releases should take place is also an important consideration. For insectivores, it is crucial that sufficient variety and abundance of invertebrate prey is available and can readily be exploited by the translocated species (Oppel and Beaven 2004). Based on my findings in Chapters 4 and 5, and the strong association of fecundity with cooler temperatures, I recommend that bandicoot releases occur in the cooler months. This recommendation is based on reproduction being energetically demanding. Therefore the increased fecundity of bandicoots in cooler weather indicates that they must be able to locate more food items due to the extended feeding times, or higher quality food items are available.

Monitoring

Traditionally, reintroduction biology has focused on determining whether reintroductions are successful (Armstrong and Seddon 2008), whilst quantitative demographic analyses of
reintroduced populations are scarce (Schaub et al. 2009). Extensive monitoring of reintroduced individuals is essential to allow for rigorous assessment of the success of reintroduction projects (Benson and Chamberlain 2007), and to assess limiting factors and predict population viability (Armstrong and Ewen 2002). This research can then be used to guide management of the population and decisions about future reintroductions (Armstrong and Ewen 2002).

Every reintroduced population of eastern barred bandicoots has been monitored by cage trapping, but this data has not often been subjected to rigorous analysis or used in an adaptive management approach. There has also been much variation in timing and frequency of monitoring events. Trapping is by far the best form of monitoring for this species, as it allows the collection of the most data. However, in the case of Mt Rothwell this technique was not practical, in terms of time and effort, due to the high capture rate of non-target species (Chapter 7), nor was it suitable for monitoring the low density population at Woodlands Historic Park (Chapter 3). Therefore, monitoring protocols should be adapted for each site individually and they should aim to collect the most useful data, which can be used to frequently analyse the population dynamics as in Chapters 4 and 5. Doing this will ensure that any population declines are detected quickly, and immediate action can then ensue to determine the cause of the decline and reverse its effects.

Given the short life span of eastern barred bandicoots I recommend that monitoring sessions should be conducted at least quarterly and preferably in the same month each quarter. This will ensure that large population fluctuations will be identified quickly and any differences between seasons can be determined. If trapping results in very few or zero captures, such as that seen at Woodlands Historic Park (Chapter 3), then valuable time and effort should focus on establishing and reversing the cause of decline, instead of implementing alternative monitoring techniques.

The importance of monitoring data is highlighted in Chapters 4 and 5, where the regulating process acting upon the Mooramong and Hamilton Community Parklands populations were determined. As more data is collected from these populations, the analyses should be updated, particularly in the case of Hamilton, which currently contains a small data set of just 10 monitoring sessions over 2.5 years (Chapter 5). This population is showing some evidence of an Allee effect. If this pattern continues as more data is obtained, then this 100-ha reserve could be too small to hold a viable self sustaining population and may need
to be managed as part of a metapopulation. Even if this is not a true Allee effect, this population cannot be harvested, to stock other reintroduction sites, at this time with no cost in terms of viability, as it does not show negative density dependent regulation (Armstrong and Ewen 2002). I recommend that a 5-year data set is probably more apt for drawing robust conclusions about the dynamics and regulatory processes of this population.
References


References


Sinclair ARE, Krebs CJ (2002). Complex numerical responses to top-down and bottom-up processes in vertebrate populations. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences*, **357**: 1221-1231.


References


Author/s: 
Winnard, Amy L.

Title: 
Reintroduction biology of the eastern barred bandicoot

Date: 
2010

Citation: 

Persistent Link: 
http://hdl.handle.net/11343/35840

File Description: 
Reintroduction biology of the eastern barred bandicoot

Terms and Conditions: 
Terms and Conditions: Copyright in works deposited in Minerva Access is retained by the copyright owner. The work may not be altered without permission from the copyright owner. Readers may only download, print and save electronic copies of whole works for their own personal non-commercial use. Any use that exceeds these limits requires permission from the copyright owner. Attribution is essential when quoting or paraphrasing from these works.