The ecology of the east-coast free-tailed bat

*(Mormopterus norfolkensis)* in the Hunter region

Anna McConville

B.Env.Sc.

This thesis has been submitted for the degree of Doctor of Philosophy with the

School of Environmental and Life Sciences

The University of Newcastle,

Australia

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Statement of Originality

I hereby certify that the work embodied in this thesis is the result of original research. The thesis contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text or co-authorship statement. I give consent to the final version of my thesis being made available worldwide when deposited in the University's Digital Repository, subject to the provisions of the Copyright Act 1968.

Signed: ............................................. Date: ....................................
Anna McConville

Signed: ............................................. Date: ....................................
Jenny Cameron
Assistant Dean (Research Training)
Statement of Collaboration and Co-authorship

I, Anna McConville, am the primary author of six co-authored papers that form this thesis. I have been responsible for the development of ideas, research design, data collection, data analysis and the writing of these papers. My supervisors Dr Bradley Law and Professor Michael Mahony assisted in the development of ideas and research design for each chapter. Dr Trent Penman provided statistical advice for the data analysis of Chapter 2 (regional habitat models). Dr Leroy Gonsalves conducted the molecular analysis of faecal material for Chapter 7 (diet). Mark Thompson collected data and contributed to the research design for the bearing error test component of Chapter 6 (home range). All of the above co-authors assisted in the review of manuscript drafts. Additional contributors are acknowledged at the end of each manuscript.

Signed: …………………………………… Anna McConville  Date: …………………..

Endorsement by Co-authors

Signed: …………………………………… Bradley Law  Date: …………………..

Signed: …………………………………… Michael Mahony  Date: …………………..
Signed: ........................................... Trent Penman  Date: .........................

Signed: ........................................... Leroy Gonsalves  Date: .........................

Signed: ........................................... Mark Thompson  Date: .........................

Endorsement by the Assistant Dean (Research Training)

Signed: ........................................... Jenny Cameron  Date: .........................
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Abstract

A detailed understanding of the ecology of threatened species is essential if we wish to develop effective conservation management strategies. *Mormopterus norfolkensis* (east-coast free-tailed bat) is a threatened insectivorous bat species of which little is known. The aim of this thesis was to address knowledge gaps regarding key aspects of the ecology of *M. norfolkensis*, including habitat, roost selection and diet.

Habitat use was investigated at multiple spatial scales, using three independent and systematically collected datasets. Overall, preferred habitat for *M. norfolkensis* was identified as productive floodplain areas, especially freshwater wetland, with urban land-use and dry sclerophyll forest avoided. Habitat use by *M. norfolkensis* was contrasted with two other morphologically similar and sympatric molossid species. Despite having similar morphologies and echolocation designs, differences in habitat use among species were found. The broad habitat types predicted from habitat models prepared at a regional-scale, using presence-absence data, were generally consistent with local-scale models, prepared using an index of activity. However, the fine-scale predictive ability of regional-scale models was poor, indicating that a cautious approach be adopted regarding their use at fine-scales, particularly when the consequences of error are severe.

In a detailed study of roost selection by a maternity colony, lactating female *M. norfolkensis* were found to be faithful to two patches of mangrove forest close to where they were captured. Females regularly switched roosts and roosted in hollows singularly or in small groups. Maternity roosts were located in locally unique mangrove forest which had abundant hollow-bearing trees and a stable microclimate. Finally, six insect orders were recorded in the diet of *M. norfolkensis*, with Lepidoptera and Diptera the most frequently encountered. A new and developing molecular method of prey identification
provided identifications to a finer taxonomic resolution, but relatively few prey taxa identifications per sample, compared to traditional microscope methods. Additionally, recommendations for conservation management and further research are provided.
Preface

This thesis is presented as a collection of published and submitted journal papers, each of which contributes to addressing the aims of this thesis. The papers are linked and integrated into the existing scientific literature by a general introduction chapter and the results summarised in a general discussion chapter. All references are included at the end of the thesis and the layout is summarised below:

Chapter 1: Introduction


Chapter 8: General Discussion
Conference presentations and posters that have relevance to the thesis but are not included:


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Chapter 1

Introduction
Chapter 1 Introduction

Bats (Order Chiroptera) are the second most diverse group of mammals on earth, comprising at least 1,116 species (Kunz and Parsons 2009). They occur in almost every ecosystem and display a remarkable range of adaptations (Kunz and Parsons 2009). Their nocturnal behaviour, small size and great mobility means that insectivorous bats are generally difficult to study and this is likely to explain the paucity of information regarding the ecology of many species. Indeed, in 2003, less than 7 % of mammal studies worldwide focused on bats (Barclay and Harder 2003). In Australia, this lack of information means that despite the presence of conservation legislation for almost 20 years, the actual conservation status of many insectivorous bat species cannot be adequately determined (Armstrong 2011, Milne and Pavey 2011, Pennay et al. 2011). Studies which have broad objectives, such as those investigating community-level relationships and functional groups are unlikely to be undertaken at a resolution to provide an insight into the ecology of rare species, which may have unusual or specialised behaviours (e.g. the spider specialist Kerivoula papuensis; Schulz 2000). The recent extinction of *Pipistrellus murrayi* (Christmas Island pipistrelle), illustrates that rigorous study of bats combined with decisive management action is urgently needed in Australia (Lunney et al. 2011a).

A basic understanding of the natural history and ecology of individual bat species is essential if we are to develop adequate conservation management strategies for Australian bats (Lunney et al. 2011a). Indeed, 45 % of the priority actions required to reverse threats to threatened insectivorous bats in New South Wales (NSW) were research-based actions (Lunney et al. 2011b). Studies that investigate some of the most fundamental components of insectivorous bat ecology, habitat use, roost selection and
diet for threatened species are particularly pressing (see The Office of Environment and Heritage 2013b, for research priorities).

1.1 Habitat selection

Habitat is where a species lives either temporarily or permanently and habitat selection refers to the behavioural mechanisms by which animals choose habitat and the evolutionary significance (adaptive advantages) of these behaviours (Krebs 2001). Along with distinguishing morphology and vocalisations, habitat is a fundamental characteristic used by ecologists to describe individual species. Habitat descriptions also form the basis of many impact assessments and conservation management strategies. However, for many species of conservation concern, preferred habitat is not well defined and the reasons for rarity may not be obvious. Highly mobile, nocturnal species, such as insectivorous bats are particularly problematic. They are often difficult to survey and their ability to traverse a range of different vegetation types means that the limited anecdotal description of habitat may at best be mis-leading or generalised and at worst result in wrong decisions on habitat protection.

The morphology of insectivorous bats, including measurements of body size and wing shape, has been related to microhabitat use (McKenzie and Rolfe 1986, O’Neil and Taylor 1986, Aldridge and Rautenbach 1987, Norberg and Rayner 1987, Crome and Richards 1988, but see Saunders and Barclay 1992, Bullen and McKenzie 2001). Additionally, the type of echolocation call used by bats is correlated with microhabitat use (Aldridge and Rautenbach 1987, Schnitzler and Kalko 2001). Species that use open and edge microhabitats ('open-space foragers' and 'edge and gap foragers' in Schnitzler and Kalko 2001) generally have low frequency and narrowband echolocation calls (Aldridge and Rautenbach 1987, Schnitzler and Kalko 2001). Furthermore, there is a trend for open- and
edge-adapted species to have higher wing loadings (a measure of wing area relative to body mass), higher aspect ratios (an index of wing shape) and to be faster flying and less manoeuvrable than clutter-adapted species (O’Neil and Taylor 1986, Aldridge and Rautenbach 1987, Norberg and Rayner 1987).

These adaptations to open microhabitats are associated with lower extinction risk (Safi and Kerth 2004) and have been proposed as features that allow bats to exploit urban (Avila-Flores and Fenton 2005, Duchamp and Swihart 2008, Jung and Kalko 2011, Threlfall et al. 2011, Hale et al. 2012, Luck et al. 2013) and agricultural (Law et al. 1999, Hanspach et al. 2012) landscapes. However, whether the morphology and echolocation call design adaptations of bats to particular microhabitats influences the selection of habitat at a landscape-scale has received little attention (but see Arlettaz 1999, Davidson-Watts et al. 2006, Nicholls and Racey 2006a for localised studies, Hanspach et al. 2012).

Increasingly, models of species habitat use (referred to throughout as “habitat models”, but also known as “species distribution models”; Guisan and Zimmermann 2000, Elith et al. 2006) and accompanying habitat suitability mapping (Jaberg and Guisan 2001, Wintle et al. 2005) are used to describe habitat for wide-ranging or lesser known species. Additionally, predictive maps derived from habitat models can be an important conservation tool, providing a basis for directing conservation effort, targeting future surveys and research, and investigating disturbance effects (Ferrier et al. 2002, Gibson et al. 2004, Greaves et al. 2006, Rhodes et al. 2006, Manning et al. 2007, Rebelo and Jones 2010, Razgour et al. 2011b). However, all models contain some level of error and the end user should consider the predictive ability of the model when deciding on its appropriate use (Franklin 2009). Indeed, habitat models and predictive maps should be accompanied by an evaluation of the level of error, uncertainty and predictive ability to ensure appropriate use (Elith et al. 2002).
Habitat selection operates at many spatial scales (Krebs 2001) and as such studies of habitat use should also consider the spatial scale at which it is undertaken to ensure it is appropriate to the subject species and intended use (Franklin 2009). Habitat models prepared across the entire distribution of a species are likely to describe only broad patterns in use, whilst models based on a single population or family group are likely to contain location-specific variables that are unable to be extrapolated to broader scales (e.g. Razgour et al. 2011b). Landscape-scale studies are useful to identify trends in habitat use by functional groups such as the use of urban areas by insectivorous bats (Avila-Flores and Fenton 2005, Hourigan et al. 2006, Threlfall et al. 2011). However, fine-scale studies of space-use such as those that investigate home range characteristics are fundamental to understanding ecology and behaviour (Kernohan et al. 2001) and provide detailed information that may be useful to manage threatened species (Adam et al. 1994, O'Donnell 2001, Johnson et al. 2007), identify population-level variation (Popa-Lisseanu et al. 2009) and to identify fine-scale threatening processes (Brearley et al. 2011). Thus, research focused on providing ecological knowledge to assist threatened species management should include habitat use studies that are undertaken at a range of spatial scales.

1.2 Roost selection

Insectivorous bats depend on day roosts to fulfil many aspects of their lifecycle, including shelter from weather, protection from predators, sites for raising young and conditions to meet thermoregulatory requirements (Kunz and Lumsden 2006). Roosts may be found in a variety of natural and man-made structures such as foliage, tree hollows, under bark, caves, rock crevices, mines, buildings and structures constructed by other animals (Kunz and Lumsden 2006). A lack of suitable roosts may limit the distribution and population
sizes of bats. In addition, the quality of maternity roosts (where females raise young) may affect reproductive success through influences on juvenile survival, growth and development (Kunz and Lumsden 2006). It is clear that the protection of habitat features that are required to fulfil so many important lifecycle functions is critical to the conservation of threatened bats and studies that investigate these requirements are valuable (Hurst and Lacki 1999, Law and Chidel 2004, Russo et al. 2004, Sedgeley 2006, Popa-Lisseanu et al. 2008, Timpone et al. 2010).

In Australia, approximately 66 % of insectivorous bats use tree hollows as roosts (Gibbons and Lindenmayer 2002). Historical land management practices have contributed to a substantial reduction in the tree hollow resources available to Australian fauna. It is estimated that land-clearing practices have contributed to a 80 - 90 % reduction in the number of hollow-bearing trees in agricultural areas (Gibbons and Lindenmayer 2002). Additionally, past forestry operations actively removed hollow-bearing trees from forests, with 82 % of Australian eucalypt forests estimated to have been logged by 1992 (RAC 1992) and depleted of hollow resources. As a result, the loss of hollow-bearing trees is recognised as a threatening process for many threatened fauna species including bats (Gibbons and Lindenmayer 2002). However, rigorous scientific data describing roosting habitat are lacking for many threatened insectivorous bat species in Australia and as such specific guidelines are difficult to develop. Furthermore, if compensatory habitat is required for threatened hollow-roosting bat species due to development proposals, there is little information to determine whether the area set aside for conservation will actually contain suitable roosting habitat for the species of concern.

Insectivorous bats have a high surface area to volume ratio, resulting from being small and having thin wing membranes resulting in high energetic demands for thermoregulation, despite being heterothermic. A number of strategies are thought to be
employed to minimise these costs such as the selection of roosts with suitable microclimates (Kerth et al. 2001, Sedgeley 2001, Ruczynski and Bogdanowicz 2005, Willis and Brigham 2005) and communal roosting (Sedgeley 2001, Willis and Brigham 2007). Pregnant and lactating females are subject to even greater energetic demands associated with reproduction as they produce milk and carry large dependent offspring (often > 30% of adult size at birth) through pregnancy and between day roosts until weaning (3 - 10 weeks; Richards 2008). As such, it is likely that breeding females face intense selection pressure to choose roosts with characteristics that are a compromise between those that minimise energy expenditure and those that maximise reproductive success. There is variation in the roost selection strategies employed by different species and these can vary seasonally (Kunz and Lumsden 2006). However, the reasons for this are poorly understood.

The benefits of sociality at roosts, such as co-operative breeding (Emlen 1991), increased foraging efficiency (Beauchamp 1999) and the detection and evasion of predators (Pulliam 1973) are constantly being traded off against the costs. These costs include direct competition for resources (Milinski and Parker 1991), increased chance of detection by predators (Vine 1973), greater susceptibility to the spread of parasites and diseases (Brown and Brown 1986) and difficulties associated with group decision-making and conflict resolution (Kerth et al. 2006). Maternity colonies of many hollow-roosting bat species have been proposed to operate as fission-fusion societies (eg Kerth and Konig 1999, O'Donnell 2000, Willis and Brigham 2004, Rhodes 2007, Popa-Lisseanu et al. 2008). In these societies, the colony consists of many small groups that are distributed among different day roosts (hereafter referred to as 'roosting groups'). The day roosts are switched regularly over time, but individuals remain faithful to the patch, so that eventually, individuals mix among the colony as a whole (Kerth and Konig 1999, Willis et al. 2003, Willis et al. 2006, Popa-Lisseanu et al. 2008). In this way, fission-fusion societies
are thought to gain the benefits of sociality, reduce parasite loads (Reckardt and Kerth 2007) and reduce predator detection, by switching roosts often (Lewis 1995).

1.3 Diet

Diet is an important component of an animal’s ecology and may provide insight into foraging behaviour, distribution, reasons for decline or rarity and possibly identify new anthropogenic threats. By comparing diet with prey availability, studies may gain an insight into the various functional and behavioural mechanisms that influence prey selection (Jones and Rydell 2003). Insectivorous bats are small, fast-flying and nocturnal and this combined with the small size of their insect prey, makes observational studies very difficult. Much of the data on insectivorous bat diet is collected from prey remains in faecal material and stomach contents (Whitaker et al. 2009). Whilst the analysis of stomach contents allows the identification of prey items prior to digestion which assists identification, animals are required to be euthanised in order to obtain these samples and this raises ethical concerns (Whitaker et al. 2009). As a result, the collection of faecal material from free-living bats has been the favoured method of studying insectivorous bat diet (Whitaker et al. 2009). However, the identification of insect prey from faecal material under a microscope is biased towards hard-bodied prey items that survive mastication and digestion (Kunz and Whitaker 1983, Dickman and Huang 1988) and identifications are usually restricted to broad taxonomic classifications such as order or family (Razgour et al. 2011a).

Recently, researchers have used molecular methods to identify insect prey items in the faecal material of insectivorous bats (Clare et al. 2009, Bohmann et al. 2011, Clare et al. 2011, Razgour et al. 2011a, Zeale et al. 2011, Alberdi et al. 2012, McCracken et al. 2012, Gonsalves et al. 2013a). DNA from prey is extracted from faecal material and sequences
compared to reference barcode libraries to identify what bats are eating. This method represents a major advance for dietary studies and is likely to become the preferred method of studying diet in small animals, such as insectivorous bats. Studies are now able to address questions that have previously remained unexplored, such as those targeting small, soft-bodied prey items, like mosquitoes (Gonsalves et al. 2013a). However, these methods are still being refined and comparisons between different techniques are particularly useful (e.g. Dodd et al. 2012).

The characteristics of echolocation calls have been related to insectivorous bat diet in a similar manner to how morphology and echolocation call design have been related to habitat use. Echolocation call design has been proposed to constrain the detection of small prey, effectively making small prey items unavailable to bats with low frequency echolocation (Mohl 1988, Barclay and Brigham 1991). Bats with low frequency echolocation calls also tend to be larger in size than species with high frequency calls (Jones 1999). However, there is much debate about the constraints imposed by echolocation on minimum detectable prey sizes as medium- to large-sized bats have been found to consume small prey (Rydell 1989, Jones 1995, Waters et al. 1995, Gonsalves et al. 2013a). Small prey may be more detectable in swarms and this may allow detection by bats with low frequency echolocation (Jones and Rydell 2003). However, some studies have proposed that echolocation call frequency is independent of detection range (Waters et al. 1995) and that call characteristics may be adjusted depending on task and habitat (Jakobsen et al. 2012).

Predator-prey interactions are an important consideration in dietary studies. Some moths and other insects have tympanic organs that provide ultrasonic hearing and an opportunity to implement a defence strategy (such as avoidance or deterrence) against predation by echolocating bats (Jones and Rydell 2003). The allotonic frequency hypothesis (Fullard
1987) predicts that bats calling within the hearing range of tympanate moths (20 - 50 kHz) will have fewer tympanate moths in their diet compared to bat species with echolocation calls outside of the hearing range of moths and this is supported by the diets of some bats (e.g. Rydell and Arlettaz 1994, Pavey and Burwell 1998). However, these predator-prey interactions are not fully understood and tympanate moths have been found in the diet of some bats with moth-audible echolocation calls using molecular methods (Clare et al. 2009, Bohmann et al. 2011).

1.4 Study species - Mormopterus norfolkensis

Mormopterus norfolkensis Gray, 1839 (east-coast free-tailed bat) is a relatively small (7 - 9 g) molossid bat, found on the east coast of Australia. It is listed as vulnerable under the New South Wales (NSW) Threatened Species Conservation Act 1995 and as vulnerable C1 under the International Union for Conservation of Nature (IUCN) red list (IUCN 2009). The species is rarely captured (only 130 reliable observations in NSW until 2007; Pennay et al. 2011) and most ecological information is based largely on anecdotal observations and limited data (e.g. Law et al. 2000, Lloyd et al. 2006, Basham et al. 2010, Threlfall et al. 2012).

Genetic data indicate that seven species of the genus Mormopterus occur in Australia (Adams et al. 1988; Reardon 2008). Whilst most species are waiting formal taxonomic revision, M. norfolkensis has long been recognised as being taxonomically distinct and has been recently redescribed (Reardon et al. 2008). In a field guide to Australian bats, Churchill (2008) used the genus Micronomus in place of Mormopterus to describe this species. However, this nomenclature has yet to receive wide acceptance and as such, we use the name Mormopterus norfolkensis throughout this thesis, as recommended by Reardon (2009).
*Mormopterus norfolkensis* has a moderately low frequency and narrowband echolocation call (characteristic frequency 31 - 35 kHz; Pennay et al. 2004), high wing loading (a measure of wing area relative to body mass; 8.87 N / m²; Rhodes 2002) and moderate aspect ratio (an index of wing shape 7.52; Rhodes 2002). These morphological and echolocation call characteristics indicate that *M. norfolkensis* is adapted to open and edge microhabitats (Aldridge and Rautenbach 1987, Norberg and Rayner 1987, Schnitzler and Kalko 2001) where it is likely to capture insect prey by aerial interception, typical of molossid bats (Schnitzler and Kalko 2001).

The factors that may influence *M. norfolkensis* habitat use have not been rigorously investigated. However, habitat for *M. norfolkensis* is generally stated as dry sclerophyll forest and woodland east of the Great Dividing Range (Duncan et al. 1999, Churchill 2008, Hoye et al. 2008), where it has been found to use flyways and upper slopes (Lloyd et al. 2006). However, it has also been recorded from a range of other habitat types, such as cleared agricultural land containing paddock trees (Law et al. 2000), wet sclerophyll forest (Hoye et al. 2008), saltmarsh (Laegdsgaard et al. 2004, Belbase 2005) and mangrove forests (G. Hoye, *pers. comm.* 2008). There are also records of the species using suburban areas (The Office of Environment and Heritage 2011), but it was notably rare in the urban and remnant bushland areas of Sydney (Basham et al. 2010, Threlfall et al. 2012).

Anecdotal reports and limited data indicate that *M. norfolkensis* roosts in tree hollows, buildings, telegraph poles and nest-boxes (Churchill 2008, Hoye et al. 2008). However, there is no empirical evidence to quantify what types of roosts are most suitable and conservation actions regarding the loss of hollow-bearing trees are only broadly defined. The radio-tracking of one individual to approximately 6 km from its roost in northern NSW
(B. Law unpublished data) indicates that *M. norfolkensis* may have large home ranges as found for morphologically similar species (12 km reported for Mormopterus species 4; Lumsden et al. 2008).

The diet of *M. norfolkensis* has not been previously documented. However, studies of North American molossid bats, have found that Lepidoptera (moths), Coleoptera (beetles) and Hemiptera (true bugs) are the most frequently consumed prey taxa overall (Whitaker et al. 1996, Lee and McCracken 2005, McWilliams 2005, Debelica et al. 2006). However, the dietary studies undertaken on Australian molossid bat species (often with limited sample sizes) suggest interspecies variation, with different combinations of insect orders preferred by different species (Vestjens and Hall 1977, Fullard et al. 1991, Milne 2006, Reside and Lumsden 2011). Following the calculations of Mohl (1988), the minimum prey size predicted to be captured by *M. norfolkensis* would be medium-sized prey of 7.8 - 8.8 mm. Furthermore, the echolocation call of *M. norfolkensis* occurs within the range of tympanate insects and as such tympanate moths may not be regularly consumed.

Threats to *Mormopterus norfolkensis* in NSW are currently considered to be loss of hollow-bearing trees, loss of foraging habitat and the application of pesticides in or adjacent to foraging areas (The Office of Environment and Heritage 2013a). While there appears to be a shift in focus from species-level to ecosystem-level conservation priorities in Australia (Lunney et al. 2011a), most of the research-based priority actions specified in Table 1-1 require species-level studies.
Table 1-1: Priority actions for *Mormopterus norfolkensis* recovery

Source: (The Office of Environment and Heritage 2013a)

<table>
<thead>
<tr>
<th>Action</th>
<th>Priority</th>
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<tr>
<td><strong>Community and land-holder liaison / awareness and/or education</strong></td>
<td></td>
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<tr>
<td>Develop and promote State-wide bat awareness programs for schools, CMAs, landholders and industry groups etc.</td>
<td>Medium</td>
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<tr>
<td><strong>Habitat management: Ongoing EIA advice to consent and planning authorities</strong></td>
<td></td>
</tr>
<tr>
<td>Prepare EIA guidelines which address the retention of hollow bearing trees maintaining diversity of age groups, species diversity, structural diversity. Give priority to largest hollow bearing trees.</td>
<td>High</td>
</tr>
<tr>
<td><strong>Habitat management: Other</strong></td>
<td></td>
</tr>
<tr>
<td>Ensure the largest hollow bearing trees, inc. dead trees and paddock trees, are given highest priority for retention in PVP assessments. Offsets should include remnants in high productivity.</td>
<td>High</td>
</tr>
<tr>
<td>Identify areas of private land that contain high densities of large hollow-bearing trees as areas of high conservation value planning instruments and land management negotiations e.g. LEP, CAPs, PVPs.</td>
<td>High</td>
</tr>
<tr>
<td>Ensure the Code of Practice for private native forestry includes adequate measures to protect large, hollow-bearing trees and viable numbers of recruit trees.</td>
<td>Medium</td>
</tr>
<tr>
<td><strong>Habitat Protection</strong></td>
<td></td>
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<tr>
<td>Promote the conservation of these private land areas using measures such as incentive funding to landholders, off-setting and biobanking, acquisition for reserve establishment or other means.</td>
<td>High</td>
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<tr>
<td><strong>Research</strong></td>
<td></td>
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<tr>
<td>Identify the effects of fragmentation in a range of fragmented landscapes i.e. the farmland/forest interface and the urban/forest interface e.g. movement and persistence across a range of fragment sizes.</td>
<td>High</td>
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<tr>
<td>Research the roosting ecology of tree-roosting bats. For example identifying the attributes of key roosts.</td>
<td>High</td>
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<tr>
<td>Research the degree of long-term fidelity to roost trees and roosting areas in order to assess their importance and the effects of their removal.</td>
<td>High</td>
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<tr>
<td>Identify important foraging range and key habitat components for this species.</td>
<td>Medium</td>
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<tr>
<td>Investigate the effectiveness of logging prescriptions.</td>
<td>Medium</td>
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<tr>
<td>Research the effect of different burning regimes.</td>
<td>Medium</td>
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<tr>
<td>Research the effectiveness of rehabilitation measures intended to increase bat populations in degraded landscapes, such as revegetating and installing bat boxes.</td>
<td>Medium</td>
</tr>
<tr>
<td>Study the ecology, habitat requirements and susceptibility to logging and other forestry practices of this little-known species.</td>
<td>Medium</td>
</tr>
<tr>
<td>Undertake long-term monitoring of populations cross tenure in conjunction with other bat species to document changes.</td>
<td>Medium</td>
</tr>
<tr>
<td>Better define species distribution through survey in coastal lowlands on- and off-reserve</td>
<td>Medium</td>
</tr>
<tr>
<td>Identify the susceptibility of the species to pesticides.</td>
<td>Medium</td>
</tr>
<tr>
<td>Quantify any benefits of local bat populations to reducing the impact of insect pests on commercial crops.</td>
<td>Low</td>
</tr>
</tbody>
</table>
1.5 Thesis Aims and Structure

The aim of this thesis is to address knowledge gaps regarding the ecology of *M. norfolkensis* in order to better inform conservation management strategies and environmental impact assessment processes in Australia. I discuss my findings in the context of current global literature and contribute to relevant theoretic debates regarding functional ecology. This thesis is presented as a collection of published and submitted journal papers, each of which contributes to addressing the aims of this thesis.

Chapter 2 uses presence-absence records obtained from ultrasonic detectors to investigate habitat use by *M. norfolkensis* at landscape and habitat element scales in relation to a range of environmental variables, such as vegetation cover and land-use. I explore whether predictions based on morphology and echolocation call characteristics, which have been previously related to microhabitat use, can predict broad-scale habitat use. Additionally, I contrast habitat use by *M. norfolkensis* with two morphologically similar, common and sympatric molossid bats, *Mormopterus* species 2 and *Mormopterus* species 4. These habitat models and predictive mapping form the basis of comparative habitat use discussions in subsequent chapters.

In Chapter 3, I begin to address the knowledge gap in relation to roost selection, a critical habitat component. I radio-track a small number of *M. norfolkensis* in two different regions of eastern Australia and document roost characteristics. This chapter illustrates the difficulty of researching rarely captured insectivorous bats and sets the scene for the major roost selection study to follow.

In Chapter 4, I use my discovery of a large *M. norfolkensis* population roosting in mangrove forest (a vegetation type that had been largely overlooked as bat habitat in
south-eastern Australia) to explore roost selection in greater detail than was possible in Chapter 3. This chapter represents the culmination of many field trips searching for locations to capture *M. norfolkensis*. In this chapter, I radio-track lactating females to maternity roosts and explore the factors that influence roost selection at the tree, patch and landscape-scale. I discuss these findings in terms of thermoregulatory requirements and sociality.

In Chapter 5, I move the focus back to habitat use and I use local-scale habitat models created using activity levels to explore habitat use surrounding the large maternity colony. I then use this new local-scale dataset to test the predictive ability of regional-scale models constructed using presence-absence data in Chapter 2.

In Chapter 6, I track the nightly movements of lactating female *M. norfolkensis* using radio-telemetry to explore the characteristics of home ranges. I rigorously quantify the bearing error associated with the telemetry method and use this to inform analyses and ensure appropriate use of results. I relate use (expressed as individual utilisation distributions) to broad habitat types within the colony range using a recent multiple linear regression technique. I also make comparisons of habitat use within individual home ranges with regional- (Chapter 2) and local-scale (Chapter 5) habitat models created using ultrasonic detectors.

In Chapter 7, I investigate the diet of *M. norfolkensis* using both traditional microscope and new and developing molecular methods of prey identification from faecal material. I compare the results of the two methods and discuss the implications of particular prey items recorded to relevant ideas regarding insectivorous bat functional ecology.
Finally, in Chapter 8, I provide a synthesis of the major findings of each chapter and discuss the conservation management implications of these findings and future directions for research. Chapters 2 - 7 of this thesis have been written as stand-alone papers for publication and as such there is some repetition in the introduction and conclusions.
Chapter 2

Contrasting habitat use of morphologically similar bat species with differing conservation status in south-eastern Australia

Anna McConville, Bradley Law, Trent Penman and Michael Mahony

Austral Ecology
Chapter 2 Contrasting habitat use of morphologically similar bat species with differing conservation status in south-eastern Australia

2.1 Abstract

The east-coast free-tailed bat *Mormopterus norfolkensis* Gray, 1839 is a threatened insectivorous bat that is poorly known and as such conservation management strategies are only broadly prescribed. Insectivorous bats that use human-modified landscapes are often adapted to foraging in open microhabitats. However, few studies have explored whether open-adapted bats select landscapes with more of these microhabitat features. We compared three morphologically similar and sympatric, molossid bats (genus *Mormopterus*) with different conservation status in terms of their association with vegetation, climate, landform and land-use attributes at landscape and local habitat element scales. We predicted that these species would use similar landscape types, with semi-cleared and low-density urban landscapes used more than forested and heavily cleared landscapes. Additionally, we explored which environmental variables best explained the occurrence of each species by constructing post-hoc models and habitat suitability maps. Contrary to predictions, we found that the three species varied in their habitat use with no one landscape type used more extensively than other types. Overall, *M. norfolkensis* was more likely to occur in low-lying, non-urban, riparian habitats with little vegetation cover. *Mormopterus* species 2 occupied similar habitats, but was more tolerant of urban landscapes. In contrast, *Mormopterus* species 4 occurred more often in cleared than forested landscapes, particularly dry landscapes with little vegetation cover. The extensive use of coastal floodplains by the threatened *M. norfolkensis* is significant
because these habitats are under increasing pressure from human land-uses and the predicted increase in urbanisation is likely to further reduce the amount of suitable habitat.

2.2 Introduction

Understanding the factors that influence habitat use by a species is fundamental to identifying threatening processes and to the development of adequate conservation strategies. However, for many species of conservation concern preferred habitat is not well defined and the reasons for rarity may not be obvious. Highly mobile, nocturnal species, such as insectivorous bats are particularly problematic. They are often difficult to survey and their ability to traverse a range of different habitat types means that the limited anecdotal evidence of habitat use is often confused or generalised. Increasingly, habitat models (Guisan and Zimmermann 2000, Elith et al. 2006) and accompanying habitat suitability mapping (Jaberg and Guisan 2001, Wintle et al. 2005) are used to classify habitat for wide-ranging or lesser known species. Predictive maps can be an important conservation tool, providing a basis for directing conservation effort, targeting future surveys and research, and investigating disturbance effects (Ferrier et al. 2002, Gibson et al. 2004, Greaves et al. 2006, Rhodes et al. 2006, Manning et al. 2007). Additionally, predictive maps can be used to contrast habitat use by different species.

The morphology of insectivorous bats, including measurements of body size and wing shape, has been related to microhabitat use (McKenzie and Rolfe 1986, O'Neil and Taylor 1986, Aldridge and Rautenbach 1987, Norberg and Rayner 1987, Crome and Richards 1988, but see Saunders and Barclay 1992, Bullen and McKenzie 2001). The type of echolocation call used by bats is also correlated with microhabitat use (Aldridge and Rautenbach 1987, Schnitzler and Kalko 2001). Species that use open and edge microhabitats ('open space foragers' and 'edge and gap foragers' in Schnitzler and Kalko
2001) generally have low frequency and narrowband echolocation calls (Aldridge and Rautenbach 1987, Schnitzler and Kalko 2001). Additionally, there is a trend for open- and edge-adapted species to have higher wing loadings (a measure of wing area relative to body mass), higher aspect ratios (an index of wing shape) and to be faster flying and less manoeuvrable than clutter-adapted species (O’Neil and Taylor 1986, Aldridge and Rautenbach 1987, Norberg and Rayner 1987).

These adaptations to open microhabitats are associated with lower extinction risk (Safi and Kerth 2004) and have been proposed as features that allow bats to exploit urban (Avila-Flores and Fenton 2005, Hourigan et al. 2006, Duchamp and Swihart 2008, Jung and Kalko 2011, Threlfall et al. 2011, Hale et al. 2012) and agricultural (Law et al. 1999, Hanspach et al. 2012) landscapes. However, whether the morphology and echolocation call design adaptations of bats to particular microhabitats influences the selection of habitat at a landscape-scale, has received little attention (but see Arlettaz 1999, Nicholls and Racey 2006a for localised studies, Hanspach et al. 2012). Vegetation clearing associated with anthropogenic land-use creates landscapes with a range of open and edge microhabitats that may benefit open- and edge-adapted species, compared to landscapes containing mainly continuous forest. However, we might also expect these benefits to decrease in heavily cleared landscapes where vegetation cover is sparse and limited edge microhabitats remain.

Whilst habitat use by a particular species is likely to be a complex response to many different factors, species that are sympatric and morphologically similar provide an opportunity to explore landscape-scale habitat use in relation to the fine-scale predictions of the foraging guild to which they are classified. Additionally, by comparing habitat use among species which have different conservation status, we may gain an insight into the factors that contribute to the rarity of a particular species.
The rare east-coast free-tailed bat *Mormopterus norfolkensis* Gray, 1839 occurs on the east coast of Australia and is listed as vulnerable under the New South Wales (NSW) *Threatened Species Conservation Act 1995* and as vulnerable C1 under the International Union for Conservation of Nature (IUCN) red list (IUCN 2009). In the Hunter region of NSW, *M. norfolkensis* occurs in sympathy with two similar species that have yet to be formally named, but that are considered to be common: the eastern free-tailed bat *Mormopterus* species 2 and southern free-tailed bat *Mormopterus* species 4 (Adams et al. 1988). These three species have low frequency and narrowband echolocation calls (characteristic frequency 29 - 35 kHz), high wing loadings (8.16 - 11.57) and moderate aspect ratios (6.98 - 7.52; Appendix A), which indicate that they are adapted to open and edge microhabitats (Aldridge and Rautenbach 1987, Norberg and Rayner 1987).

The first aim of this study was to explore how these open- and edge-adapted bat species were distributed in relation to landscape-scale vegetation cover and land-use, and also in relation to habitat elements that are more indicative of microhabitat use. We predicted that: 1) each *Mormopterus* species would occur more frequently in semi-cleared landscapes, which lie between the extremes of forest (where there is limited edge or open microhabitats for foraging) and heavily cleared landscapes (where vegetation structure such as forest remnant edges is mostly absent); and 2) that the low density urban landscapes that occurred within the study area would provide habitat for all three species. Secondly, we aimed to explore what other factors influenced the occurrence of the rare *M. norfolkensis* and contrasted these factors with the two common species (*M. species 2* and *M. species 4*), predicting that *M. norfolkensis* would have a more restricted distribution based on habitat suitability maps.
2.3 Methods

2.3.1 Study species

Genetic data indicate that seven species of the genus *Mormopterus* occur in Australia (Adams et al. 1988, Reardon et al. 2008). However, most species are awaiting formal taxonomic revision. Within the study area, three species occur in sympatry, *M. norfolkensis*, *M. species 2* and *M. species 4* (A. McConville unpublished data, 2012; Churchill 2008, The Office of Environment and Heritage 2011). *Mormopterus norfolkensis* has long been recognised as being taxonomically distinct and has been recently redescribed (Reardon et al. 2008). The other two species, whilst recognised as distinct species (Adams et al. 1988, Reardon et al. 2008), have yet to be formally described and this paper follows the nomenclature of Adams et al. (1988). The three species are morphologically similar, but can be differentiated using a combination of features such as penis morphology and forearm length (Churchill 2008, Reardon et al. 2008, Reside and Lumsden 2011) and they have subtly different echolocation calls. See Appendix A for relevant morphometric measurements.

2.3.2 Study area

The study area is located in the Hunter and Central Coast regions of NSW on the east coast of Australia (32°56'S, 151°47'E; Figure 2-1). These regions are characterised by coastal lakes, estuaries and broad river floodplains which are bounded by steeper sandstone escarpments and mountains. The region experiences a warm temperate climate (average monthly temperatures 3.8 - 31.7 °C in the west with milder conditions nearer to the coast) and average annual rainfall of 645 mm in the west and 1096 mm at the coast (Bureau of Meteorology 2012). The mountains of the Great Dividing Range that form a barrier between eastern and western NSW are relatively low along the broad
Hunter River valley. This topography allows a mixing of dry and mesic adapted flora and fauna species along the valley floor (Peake 2006).

The floodplains within the study area consist of relatively rich alluvial soils with the hills and sandstone escarpments of poorer soil quality (McManus et al. 2000). The most common land-uses in the Hunter Valley (western part of the study area) are cattle grazing and coal mining (both underground and open cut), with low density urban centres. Coastal areas have the greatest urban development and these areas are characterised by sprawling, low density and mostly leafy suburbs, with high density development generally restricted to the major city centres of Newcastle and Gosford. Remnant native vegetation consists of eucalypt-dominated dry sclerophyll forest and woodland in the valley tending towards moister sclerophyll forest, coastal heath, wetland, saltmarsh and mangrove forest nearer to the coast. Most of the large forested areas occur in the elevated ranges.
Figure 2-1: Study area location, sample landscapes, vegetation and urban land-use cover

Study area (dashed line); major cities (black circle); ocean and major waterbody (dark grey); urban land-use (mid grey); vegetation cover (light grey); and sample landscape (black square). Four different types of landscape (5 x 5 km) were sampled: A – cleared (<10% vegetation); B – semi-cleared (20 – 40% vegetation); C – forested (>80% vegetation); and D – urban (30 - 60 % urban land-use).

2.3.3 Study design and site selection

We stratified sampling at two spatial scales, the ‘landscape’ and the ‘habitat element’ (which were sites nested within each landscape), to explore the influence of vegetation cover and urban land-use on the presence of the three Mormopterus species. A 5 km x 5 km grid was placed over the study area (770 000 ha) to delineate ‘landscapes’ (Threlfall et
al. 2011). The size of the landscape grid selected represents a large portion (approximately half) of the maximum nightly movement of the Mormopterus species' under investigation (12 km reported for M. species 4; Lumsden et al. 2008). The boundary of the study area was determined by the known distribution of M. norfolkensis and the availability of regional vegetation mapping datasets (House 2003, Peake 2006). Each landscape cell was then classified into the following categories: urban (30-60 % urban land-use and 20 - 40 % vegetation cover); cleared (< 10 % vegetation cover, < 30 % urban land-use); semi-cleared (20 - 40 % vegetation cover and < 30 % urban land-use); and forested (> 80 % vegetation cover, < 30 % urban land-use). Percent vegetation cover and percent urban land-use were calculated for each landscape cell using ArcGIS (version 9.0, ESRI, Redlands, CA, USA), a geographic information system (see Appendix C for further details).

Thirty landscape cells were selected: eight cleared, eight forested, eight semi-cleared and six urban (Figure 2-1). They were chosen randomly, irrespective of land ownership, could not be located next to each other and could not contain > 10 % major waterbody (i.e. ocean, lake). Within each landscape cell, sampling was undertaken at four sample sites (habitat elements) selected in the field to represent the main habitats within each landscape: open (> 100 m from a tree and riparian element); paddock tree (solitary tree or a group of 2-4 trees, > 100 m from a forested remnant and riparian element); forest remnant (> 15 ha) along a track or opening within a patch of forest, > 100 m from the forest edge and a riparian element); and riparian (< 10 m from a watercourse). Riparian habitat elements sampled were located on creeks and rivers (sometimes at dams on small creeks) and in cleared and semi-cleared landscapes they often contained some remnant vegetation. Open and paddock tree habitat elements did not occur in forested landscapes. Overall, a total of 98 sites were sampled: 19 open, 20 paddock tree, 29 forest remnant and 30 riparian habitat elements.
2.3.4 Bat call sampling

Sampling was undertaken between November 2007 and April 2008, which encompasses the maternity, weaning and post-reproductive period for bats. One bat detector (Anabat II with CFZCAIM, Titley Electronics, Balina) was placed at each habitat element site to passively record for two entire nights (15 sites were sampled for one night only). Surveys were not conducted on windy or rainy nights or on a full moon to avoid any possible effect of these factors on bat activity. The sensitivity setting of each detector was recorded and the detection volume was derived for each detector using AnaVolumes (Corben 2009).

Bat calls were manually identified using AnalookW (Corben 2008) to view calls and extract parameters. A key to the identification of Mormopterus species in the study area was developed to make the process less subjective (see Appendix B). The key was based on a broader NSW guide (Pennay et al. 2004) in combination with a review of reference calls and recent Mormopterus species distributional information from trapping records within the study area (A. McConville unpublished data, 2012).

Each pass (call sequence) was assigned to one of three categories - definite, probable or unidentified, according to the confidence with which identification could be made (Mills et al. 1996). Passes that did not contain any search phase pulses or contained less than three pulses were not considered. Definite and probable identifications were included in the analyses. Average nightly bat activity (number of passes) was calculated for each species at each site. As each Mormopterus species occupied a relatively small proportion of sites and activity varied between narrow limits, species presence at a site was used as the response variable.
2.3.5 Habitat models

Seventeen environmental variables were selected for analysis of habitat use and were collected at both a site- and landscape-scale (see Appendix C for variable descriptions). Landscape variables used in the analysis were derived from various spatial layers within a series of nested concentric circular buffers (0.25, 1, 5, 10 km) created around each site in ArcGIS. The maximum buffer distance of 10 km was selected as radio-tracking studies have found that the maximum distance travelled in a single night by *M. norfolkensis* was 10 km (Chapter 6) and by *M. species 4* was 12 km (Lumsden et al. 2008).

Generalised linear mixed models (GLMM’s) were used to model bat species presence / absence against environmental variables. Mixed models account for the correlation that exists in data when units are repeatedly measured. A random intercept (landscape cell identification code) was included to account for the repeated measures of each landscape cell. All other variables were treated as fixed effects. All statistical analyses were carried out in R (R Development Core Team 2008) using the lme4 (Bates et al. 2008), AICcmodavg (Mazerolle 2012), verification (NCAR - Research Application Program 2012) packages. Two main models were prepared for each species, the first being based on the *a priori* experimental design of landscape and habitat elements and the second being the best fitting model of all additional site and environmental variables (a post-hoc approach). Model residuals were checked for spatial autocorrelation using Moran’s I (Moran 1950) and all results are reported as means ± standard error.

The *a priori* landscape type and habitat element design (species presence ~ landscape type + habitat element) was input into a GLMM for each *Mormopterus* species. This may have resulted in slightly over parameterised models for species with few presence records (i.e. *M. species 2*).
For post-hoc models, we reduced the number of variables for model selection by excluding temperature, humidity, season and detector volume from the analyses as they showed no relationship with the presence of any of the Mormopterus species (Pearson correlation coefficient \( r < 0.4 \)) during preliminary exploration. To further reduce the number of candidate variables, for each species initial univariate GLMM's were constructed for each explanatory variable. We then ranked these univariate models by Akaike Information Criterion value corrected for small sample size (AICc; Akaike 1973), with the highest ranking model having the lowest AICc. A small number of explanatory variables were then selected from the highest ranking univariate models for consideration in the final candidate models, the number of variables being six for \( M. \) norfolkensis, four for \( M. \) species 2 and six for \( M. \) species 4. Where landscape buffer variables of the same category were high ranking, only the highest-ranking buffer value was selected for consideration in the final model (Burnham and Anderson 2002). To avoid collinearity in the models, where variables were highly correlated \( (r > 0.7) \), the variable with the lower ranking univariate model was excluded from analysis.

Final candidate models were then constructed with the maximum number of variables included being related to the number of presence sites in the dataset for each species. We followed the recommendation of Wintle et al. (2005) by including a maximum of one variable per 10 presence sites in the final candidate models, being three for \( M. \) norfolkensis, two for \( M. \) species 2 and four for \( M. \) species 4. Candidate models included all possible variable combinations up to the maximum number of variables allowable for each species. We then calculated AICc and model weight \( (W_i; \text{Burnham and Anderson 2002}) \) for each of the final candidate models. Relative importance for each variable was quantified through an index constructed by summing \( W_i \) for all models within the 95 % confidence set (Burnham and Anderson 2002). The best-fitting model for each species
was considered to have the lowest AICc ranking and any model within two AICc points was considered to have strong support (Burnham and Anderson 2002).

A probability of occurrence map was prepared for each of the post-hoc habitat use models with strong support using the logit transform. To provide a habitat map representative of overall habitat suitability for models which contained the NPP variable, we averaged again the average annual monthly NPP values for the warm months insectivorous bats are most active (September to March). The maps for each species were then compared using a correlation matrix (Band Collection Statistics tool in ArcGIS).

The receiver operating characteristic (ROC) was used to discrimination ability of each model measured by the area under the curve (AUC) with 0.5 representing an entirely random model (Pearce and Ferrier 2000). The traditional academic point system (Swets 1988) was used as a rough guide for classifying the discrimination ability of each model with AUC values (Thuiller et al. 2003) where AUC values under 0.7 are considered poor, values of 0.7 to 0.8 are rated as fair, 0.8 to 0.9 as good and those over 0.9 as excellent. Discrimination ability for each model was tested against both the original dataset and against a validation dataset created from regional database records from ‘The Atlas of NSW Wildlife’ (The Office of Environment and Heritage 2011). We excluded records from the validation dataset that had less than 100 m positional accuracy, were pre-2004, had less confident call identification classifications and that were within 100 m of another record. After filtering, a total of 204 presence records for *M. norfolkensis*, 145 for *M.* species 2 and 22 for *M.* species 4 were used to create the validation dataset. Random points of the same number of presence records for each species were used as pseudo-absences in the validation dataset.
2.4 Results

A total of 532 *Mormopterus* species passes were recorded (200 from *M. norfolkensis*, 93 from *M. species 2* and 239 from *M. species 4*). Activity levels at each site during the study were mostly low with 1.07 ± 3.71 passes / night (range 0 - 35) for *M. norfolkensis*, 0.48 ± 1.56 passes / night (range 0 - 11) for *M. species 2* and 1.35 ± 2.48 passes / night (range 0 - 13) for *M. species 4*. *Mormopterus* species 4 was the most widespread species during the study, occurring at 43.9 % of sites, with *M. norfolkensis* and *M. species 2* occurring less frequently (34.7 % and 23.5 %, respectively). All three *Mormopterus* species were recorded at all habitat elements sampled and all species were recorded in all landscape types sampled, except for *M. species 4* which was not recorded in any forested landscapes (Figure 2-2).
Figure 2-2: Average nightly bat activity

Average nightly bat activity (passes per night) recorded for each species in the study region across the four landscape categories (a); and four habitat elements (b). *Mormopterus norfolkensis* (Mono); *M. species 2* (Msp2); and *M. species 4* (Msp4).

### 2.4.1 A priori design models

*Mormopterus norfolkensis* was less likely to occur in urban landscapes when compared with cleared land ($p = 0.048$) and more likely in riparian habitat elements compared with open elements ($p = 0.028$; Table 2-1). Discrimination ability to the original dataset was good (AUC = 0.87). The *a priori* GLMM for *M. species 2* found that it was more likely to occur in riparian elements than the open elements ($p = 0.076$; Table 2-1), and no landscape types were found to have a significant relationship with the presence of *M. species 2*. Whilst the discrimination ability to the original dataset was good (AUC = 0.87), due to the fewer presence sites recorded for *M. species 2* it is likely to be over parameterised and as such the overall confidence in this model is less than for the other species. *Mormopterus* species 4 was found to be less likely to occur in forested landscapes than cleared landscapes ($p = 0.007$) and less likely to occur in urban areas than open landscape types ($p = 0.016$; Table 2-1). Discrimination ability to the original dataset was excellent (AUC = 0.94). There was no spatial autocorrelation in the residuals for any of the *a priori* design models ($p > 0.216$).
Table 2-1: Summary from a priori habitat models

Summary from generalised linear mixed model (GLMM) analysis of the a priori experimental design GLMM (landscape type and habitat element) from 98 sites. The landscape cell ID was added as a random effect term in the model. The referent category for landscape type is cleared and the referent category for habitat element is open. Area under the curve (AUC) values were calculated for the receiver operating characteristic based on the dataset used to create the model (training).

<table>
<thead>
<tr>
<th>Species</th>
<th>Parameter</th>
<th>Estimate ± SE</th>
<th>z value</th>
<th>p value</th>
<th>AUC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mormopterus norfolkensis</td>
<td>(Intercept)</td>
<td>-0.70 ± 0.71</td>
<td>-0.98</td>
<td>0.326</td>
<td>0.87</td>
</tr>
<tr>
<td>Landscape type</td>
<td>Forested</td>
<td>-1.42 ± 0.94</td>
<td>-1.52</td>
<td>0.129</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Semi-cleared</td>
<td>0.14 ± 0.73</td>
<td>0.19</td>
<td>0.846</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Urban</td>
<td>-1.90 ± 0.96</td>
<td>-1.98</td>
<td>0.048*</td>
<td></td>
</tr>
<tr>
<td>Habitat element</td>
<td>Forest remnant</td>
<td>-0.29 ± 0.75</td>
<td>-0.39</td>
<td>0.699</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Paddock Tree</td>
<td>0.02 ± 0.76</td>
<td>0.03</td>
<td>0.979</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Riparian</td>
<td>1.59 ± 0.73</td>
<td>2.19</td>
<td>0.028*</td>
<td></td>
</tr>
<tr>
<td>Mormopterus species 2</td>
<td>(Intercept)</td>
<td>-1.73 ± 0.82</td>
<td>-2.13</td>
<td>0.033</td>
<td>0.87</td>
</tr>
<tr>
<td>Landscape type</td>
<td>Forested</td>
<td>-0.60 ± 1.04</td>
<td>-0.58</td>
<td>0.563</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Semi-cleared</td>
<td>0.21 ± 0.85</td>
<td>0.24</td>
<td>0.808</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Urban</td>
<td>-0.05 ± 0.94</td>
<td>-0.05</td>
<td>0.961</td>
<td></td>
</tr>
<tr>
<td>Habitat element</td>
<td>Forest remnant</td>
<td>-0.73 ± 0.92</td>
<td>-0.80</td>
<td>0.426</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Paddock Tree</td>
<td>0.06 ± 0.86</td>
<td>0.07</td>
<td>0.944</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Riparian</td>
<td>1.34 ± 0.76</td>
<td>1.77</td>
<td>0.076</td>
<td></td>
</tr>
<tr>
<td>Mormopterus species 4</td>
<td>(Intercept)</td>
<td>0.79 ± 0.87</td>
<td>0.92</td>
<td>0.359</td>
<td>0.94</td>
</tr>
<tr>
<td>Landscape type</td>
<td>Forested</td>
<td>-4.59 ± 1.71</td>
<td>-2.68</td>
<td>0.007*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Semi-cleared</td>
<td>-0.33 ± 1.01</td>
<td>-0.32</td>
<td>0.748</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Urban</td>
<td>-2.91 ± 1.21</td>
<td>-2.40</td>
<td>0.016*</td>
<td></td>
</tr>
<tr>
<td>Habitat element</td>
<td>Forest remnant</td>
<td>-0.25 ± 0.77</td>
<td>-0.32</td>
<td>0.746</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Paddock Tree</td>
<td>0.36 ± 0.76</td>
<td>0.46</td>
<td>0.648</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Riparian</td>
<td>0.47 ± 0.77</td>
<td>0.61</td>
<td>0.545</td>
<td></td>
</tr>
</tbody>
</table>

* Significant differences reported at α = 0.05.
2.4.2 Post-hoc models

*Mormopterus norfolkensis* was more likely to occur in areas which had less vegetation within 1 km, were lower in elevation and did not have urban land-use within 10 km (Figure 2-3) in the best fitting post-hoc model (MnModel A; Appendix D). The 95% confidence set of models for *M. norfolkensis* contained 29 models, indicating a high level of model uncertainty. However, the variables with the highest relative importance indices (Table 2-2) were contained within the best fitting post-hoc model. ROC plots for the original dataset indicated discrimination ability was excellent (AUC = 0.90). However, ROC plots for the validation dataset indicated that discrimination ability to regional species records was poor (AUC = 0.58). A total of five other models had strong support (MnModels B - F; Appendix D). Low elevation and lack of urban development within 10 km were common in five out of six of the best fitting models. The habitat map (Figure 2-4a) for the best fitting model for *M. norfolkensis* was correlated (r = 0.64 - 0.86) to maps created from supported models. Of these maps, models containing the landform factor (MnModelD and MnModelF) were the least well correlated (r = 0.64 and r = 0.76 respectively), which is likely to be due to the finer topographic detail contained within the landform layer. The full post-hoc model results are provided in Appendix D.
Table 2-2: Relative importance indices for post-hoc habitat models

Relative importance indices (Burnham and Anderson 2002) calculated from AICc values of the 95% confidence set of post-hoc GLMMs for Mormopterus norfolkensis (29 models), M. species 2 (five models) and M. species 4 (26 models). Highest value for each species is highlighted in bold. Variables that were not included in the 95% confidence model set are indicated by a dash. See Appendix C for full description of factors.

<table>
<thead>
<tr>
<th>Factor</th>
<th>M. norfolkensis</th>
<th>M. species 2</th>
<th>M. species 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average annual rainfall</td>
<td>-</td>
<td>-</td>
<td>0.55</td>
</tr>
<tr>
<td>Low canopy height</td>
<td>-</td>
<td>-</td>
<td>0.32</td>
</tr>
<tr>
<td>Elevation (log transformed)</td>
<td>0.51</td>
<td>0.72</td>
<td>-</td>
</tr>
<tr>
<td>Landform</td>
<td>0.33</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Nearest riparian land-use (log transformed)</td>
<td>0.26</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Net plant productivity within 1 km</td>
<td>-</td>
<td>0.56</td>
<td>-</td>
</tr>
<tr>
<td>Net plant productivity within 5 km</td>
<td>-</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td>Presence of riparian land-use within 1 km</td>
<td>-</td>
<td>0.37</td>
<td>-</td>
</tr>
<tr>
<td>Presence of riparian land-use within 5 km</td>
<td>-</td>
<td>-</td>
<td>0.2</td>
</tr>
<tr>
<td>Slope (log transformed)</td>
<td>0.25</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Presence of urban land-use within 1 km</td>
<td>-</td>
<td>0.25</td>
<td>-</td>
</tr>
<tr>
<td>Presence of urban land-use within 10 km</td>
<td>0.58</td>
<td>-</td>
<td>0.32</td>
</tr>
<tr>
<td>Vegetation area within 1 km</td>
<td>0.58</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Vegetation area within 10 km</td>
<td>-</td>
<td>-</td>
<td>0.89</td>
</tr>
</tbody>
</table>

The best fitting post-hoc model for M. species 2 (Msp2Model A; Appendix D) found M. species 2 was more likely to occur in areas at lower elevation with higher productivity (Figure 2-3). The 95% confidence model set for M. species 2 contained five models, indicating some minor model uncertainty and the best fitting model was considered to have a fair discrimination ability using both original and validation dataset ROC plots (AUC = 0.73 and 0.71, respectively). There were three supported models (Msp2Models B - D; Appendix D). Model weights indicated that elevation was the most important variable with
a relative importance index of 0.72 (Table 2-2). Whilst the presence of riparian and urban land-uses within 1 km variables were included in supported models, they were of lower importance (relative importance indices of 0.37 and 0.25 respectively; Table 2-2). Habitat maps of supported models for *M.* species 2 were correlated with the map of the best-fitting model ($r > 0.78$; Figure 2-4b). However, maps for the supported models were not well correlated to each other ($r = 0.50$), which may be due to the statistical noise from the inclusion of less important variables or that some influential variables were not included.

*Mormopterus* species 4 was more likely to occur in landscapes with less vegetation and lower average annual rainfall in the best fitting *post-hoc* model (Msp4ModelA; Figure 2-3; Appendix D). The 95% confidence set of models for *M.* species 4 contained 26 models, indicating a high level of model uncertainty. However, the two variables with the highest overall model weights, vegetation within 10 km (relative importance index = 0.89) and average annual rainfall (relative importance index = 0.55; Table 2-2) were contained within the best fitting *post-hoc* model. Furthermore, all of the supported models for *M.* species 4 (Msp4Model B - E and Msp4Model X; Appendix D) contained a negative relationship with the vegetation within 10 km variable indicating that the best fitting model was representative of the 95% confidence model set. Low canopy height was included as a non-significant variable in one of the strongly supported models (Msp4Model X; Appendix D) and this model could not be mapped as low canopy height was not available as a GIS layer. Habitat maps for each of the supported models for *M.* species 4 were highly correlated with the best fitting model ($r > 0.97$; Figure 2-4c) most likely due to the consistent presence of the vegetation within 10 km factor. There was no evidence of spatial autocorrelation in the residuals for any supported model for any of the study species ($p > 0.189$).
Figure 2-3: *Post-hoc* habitat model response plots

Partial-plots of the relationship between probability of occurrence and environmental variables for the best-fitting *post-hoc* models for *Mormopterus norfolkensis* (Mono); *Mormopterus* species 2 (Msp2); and *Mormopterus* species 4 (Msp4). The dashed lines indicate 95% confidence intervals. The x-axis represents the range of values sampled for each environmental variable. Over-plotting of multiple points is visualised by increasingly darker shades of grey.
Figure 2-4: Probability of occurrence maps

Probability of occurrence maps for *Mormopterus norfolkensis*, *M*. species 2 and *M*. species 4, created from the best-fitting post-hoc generalised linear mixed model of landscape variables from 98 sites. The landscape cell ID was added as a random effect term in each model. Models were: *M. norfolkensis* ~ Vegetation within 1km + Elevation (log transformed) + Urban land-use presence within 10km; *M*. species 2 ~ Net Primary Productivity within 1km + Elevation (log transformed); *M*. species 4 ~ (Vegetation within 10km / 100) + Average Annual Rainfall.

2.5 Discussion

Contrary to predictions based on ecomorphology, but consistent with previous studies of similar sympatric species (Arlettaz 1999, Nicholls and Racey 2006a), the three morphologically similar *Mormopterus* species varied in their habitat use and regional maps indicated different distributions, with the rare *M. norfolkensis* not being the most restricted.
There was also no clear trend for these species to select landscapes that have more open and edge microhabitats (semi-cleared landscapes, prediction 1). Most importantly, *M. norfolkensis* avoided urban areas, contrary to prediction 2, whereas these areas were used by the common *M.* species 2 and *M.* species 4. *Mormopterus norfolkensis* was also more likely to occur in low-lying riparian areas that are subject to increasing anthropogenic pressures which further supports conservation concerns regarding the species.

### 2.5.1 Limitations

Whilst most models had very good to excellent model fit to the original dataset, when model fit was calculated against the validation dataset it was poor, being little better than random for most models. The models were created using a dataset specifically collected for this study to avoid the potential sources of bias and error associated with regional species databases such as species misidentification, positional accuracy, temporal variation and bias towards population centres or conservation reserves. However, these potential biases and errors are still present within the regional database records used to create the validation dataset and may be responsible for the poor model fit. It is also possible that the use of presence / absence data in the modelling process may have resulted in commuting bats being given a higher status and over-predicting habitat, compared to modelling bat activity levels. We suggest that a realistic discrimination ability value lies somewhere between the AUC calculated from the unreliable validation dataset and the optimistic AUC calculated from the original dataset. Until more detailed models can be developed and tested using other approaches such as activity levels, variable interactions or by informative indexing of environmental variation, we suggest that the current models will be most useful for describing broad patterns of use.
2.5.2 Habitat use

The response to landscape vegetation cover varied among species. Landscapes with less vegetation cover were important in predicting the occurrence of *M*. species 4, with vegetation cover included as a factor in all models. However, the amount of landscape vegetation cover was less consistent in predicting the occurrence of *M. norfolkensis* and *M*. species 2. The occurrence of *M*. species 2 was not associated with vegetation cover in any of the models and whilst *M. norfolkensis* was found to be more likely to occur in landscapes with less vegetation cover in the best fitting post-hoc model, other factors were also highly influential. Additionally, contrary to prediction 1, the intermediate category of semi-cleared landscapes was not significantly associated with the occurrence of any of the *Mormopterus* species in *a priori* models indicating that the amount of edge habitat at a landscape-scale may not be important.

The habitats mapped for *M*. species 2 and *M. norfolkensis* were generally similar, being low-lying major riparian areas. Riparian elements were significant factors in *a priori* models for both *M. norfolkensis* and *M*. species 2 and whilst the post-hoc models did not contain a riparian land-use factor, predictive mapping suggests that these areas are suitable. The riparian habitat elements sampled during this study often contained scattered trees and remnant vegetation and it is possible that bats were selecting for these edge microhabitats in cleared and semi-cleared landscapes. However, the paddock tree habitat elements, which also represent edge microhabitat, were infrequently used, suggesting that edges alone are unlikely to explain the frequency of riparian habitat element use. Riparian habitats often support high levels of bat activity, which may be due to high insect prey abundance (Racey and Swift 1985, Hayes 1997, Grindal et al. 1999). Additionally, more productive soils have been found to be related to high bat activity levels (Law et al. 2011a, Threlfall et al. 2011). In the case of *M. norfolkensis*, vegetation cover could be an inverse surrogate for productive environments. The more fertile floodplains
within the study area have been historically cleared for agriculture and extensive productive wetlands (in a natural state and used as pasture) were not included in the ‘woody’ vegetation classification used and so were considered to be ‘cleared’.

*Mormopterus* species 4 showed no preference for riparian habitats, and this is consistent with other studies (Law and Chidel 2006, Law et al. 2011b, Reside and Lumsden 2011).

Paddock trees in agricultural areas of Australia are important to insectivorous bats when compared with cleared habitats and remnant forest patches (Law et al. 2000, Lumsden and Bennett 2005, Fischer et al. 2010a, Fischer et al. 2010b, Hanspach et al. 2012). However, none of the species in our study were strongly associated with paddock tree habitat elements. These species are highly mobile and would not be limited by the relative isolation of paddock trees. Explanations for paddock trees being used by insectivorous bats include high insect abundance (Law et al. 2000, Lumsden and Bennett 2005) and a point of navigation by having an object to reflect echoes in cleared landscapes. However, perhaps productivity is a better predictor of occurrence than the presence of paddock trees alone. For example, bats may occur at paddock trees in productive landscapes, but may be less likely to occur in regions of poorer fertility, irrespective of whether paddock trees are present.

The influence of urban land-use on habitat use was also a point of contrast among species. The rare *M. norfolkensis* avoided urban landscapes, *M.* species 2 used them somewhat and urban land-use was a relatively unimportant factor in predicting the occurrence of *M.* species 4. Previous studies found that *M.* species 2 was relatively widespread and common in urban areas compared to other bat species (Hourigan et al. 2006, Basham et al. 2010, Reside and Lumsden 2011, Threlfall et al. 2012). However, within our study area, the low density urban areas appear to represent only marginal
habitat for *M*. species 2, with the most suitable habitats (> 0.5 probability of occurrence), generally mapped outside of urban areas.

Our findings are consistent with a previous study that suggested *M. norfolkensis* was moderately sensitive to urbanisation (Threlfall et al. 2012). However, it is unclear why *M. norfolkensis* avoids urban landscapes. The low roost availability in urban habitats is unlikely to influence foraging use by *M. norfolkensis* as it can travel up to 10 km to forage each night (Chapter 6). Similarly, the abundance of prey orders that dominate the diet of *M. norfolkensis*, (Lepidoptera and Diptera; Chapter 7), were not significantly different between urban and natural habitats in a nearby study (Gonsalves et al. 2013b). It is possible that a subtle difference in morphologies among the species explains the difference in habitat use. It is also possible that competitive interactions occur between *M*. species 2 and the slightly larger *M. norfolkensis* in coastal areas. Future studies that investigate morphologically similar and sympatric species would benefit from the collection of detailed air frame measures, such as those presented by Bullen and McKenzie (2001). Additionally, studies that relate various morphological measures to habitat use at a range of spatial scales and those that investigate inter-species interactions and response to human disturbance (noise and artificial lighting) would be valuable.

This is the first study to successfully model habitat for the threatened *M. norfolkensis* and these models will provide the basis for future ecological research and conservation effort. Our study illustrates that it is difficult to make detailed predictions of broad-scale habitat use based on coarse-grained morphological adaptations and echolocation call design characteristics, as habitat selection is likely to be a result of many different competing factors.
2.6 Acknowledgements

We would like to thank the numerous property owners and managers that generously allowed access for sampling and the volunteers who assisted with field work. Caragh Threlfall and three anonymous referees provided helpful comments that greatly improved this manuscript. Funding for this PhD project was sourced from: The Tom Farrell Institute for the Environment; Donaldson Conservation Trust; Hunter-Central Rivers Catchment Management Authority; Royal Zoological Society of NSW (Ethel Mary Read Award); Wambo Coal Pty Ltd; and Australian Geographic Society.
**Appendix A: Study species morphology**

**Table A1: Morphological and call frequency measures**

Morphological and call frequency measures for *Mormopterus norfolkensis*, *M*. species 2 and *M*. species 4 obtained from literature (sources indicated by footnotes).

<table>
<thead>
<tr>
<th>Measures</th>
<th><em>Mormopterus norfolkensis</em></th>
<th><em>Mormopterus species 2</em></th>
<th><em>Mormopterus species 4</em></th>
</tr>
</thead>
<tbody>
<tr>
<td>Forearm (mm)</td>
<td>37.2 (16) <em>a</em></td>
<td>33.1 (31) <em>h</em></td>
<td>33.8 (100) <em>b</em></td>
</tr>
<tr>
<td>Mass (g)</td>
<td>8.6 (20) <em>c</em></td>
<td>10.0 (31) <em>h, d</em></td>
<td>9.2 (100) <em>b</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>7.5 (17) <em>c, d</em></td>
<td></td>
</tr>
<tr>
<td>Wing span (m)</td>
<td>0.269 (20) <em>c</em></td>
<td>0.232 (31) <em>b</em></td>
<td>0.233 (100) <em>b</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.255 (17) <em>c</em></td>
<td></td>
</tr>
<tr>
<td>Wing area (m²) <em>e</em></td>
<td>0.010 (20) <em>c</em></td>
<td>0.007 (31) <em>b</em></td>
<td>0.008 (100) <em>b</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.009 (17) <em>c</em></td>
<td></td>
</tr>
<tr>
<td>Wing loading <em>e</em></td>
<td>8.86 (20) <em>c</em></td>
<td>13.70 (31) <em>b</em></td>
<td>11.57 (100) <em>b</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>8.16 (17) <em>c</em></td>
<td></td>
</tr>
<tr>
<td>Aspect ratio <em>e</em></td>
<td>7.52 (20) <em>c</em></td>
<td>7.50 (31) <em>b</em></td>
<td>6.98 (100) <em>b</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td>7.23 (17) <em>c</em></td>
<td></td>
</tr>
<tr>
<td>Echolocation call frequency (kHz) <em>f</em></td>
<td>31 - 35 (15)</td>
<td>29 - 32 (5)</td>
<td>26 - 31 (30)</td>
</tr>
<tr>
<td>Echolocation call shape <em>e</em></td>
<td>Quasi-constant frequency</td>
<td>Quasi-constant frequency</td>
<td>Quasi-constant frequency</td>
</tr>
</tbody>
</table>

Values are presented as mean (sample size) for all measurements, except for the characteristic frequency of echolocation calls where the range (sample size) is given. Wing loading was calculated as mass (g) times gravitational force (9.81 m/s²) and divided by wing area (m²). Aspect ratio was calculated by squaring wing span (m) and then dividing by wing area (m²).

* *a* Sourced from Reardon et al. (2008)*

* *b* Adapted from Reside and Lumsden (2011). Unit conversions were undertaken prior to recalculation of wing loading and aspect ratio: wing area - cm² to m²; mass - g to kg; and wing span from cm to m

* *c* Sourced from Rhodes (2002)*

* *d* Note: there is a difference in the average weight of *M*. species 2 between studies which is important to calculations of wing loading and aspect ratio. The average weight of *M*. species 2 is 0.009 kg (n = 72; Churchill 2008) which falls between the values reported here

* *e* Note: the methods used to measure wing area differed slightly. Rhodes (2002) included the head in the wing area and Reside and Lumsden (2011) did not. As wing area is used in the calculation of both wing loading and
aspect ratio, these measures would be slightly higher using the method of Rhodes (2002) than if the same individual was measured using the method of Reside and Lumsden (2011).

* Extracted from reference calls collected by AM within the study area and from a regional bat call identification guide (Pennay et al. 2004)
Appendix B: Echolocation call identification key

The echolocation calls recorded during the study were identified using a key adapted from a regional bat call identification guide (Pennay et al. 2004). A call sequence (pass) is made up by multiple sound pulses made by an echolocating bat as it flies by the bat detector. Each pass was assigned to one of three categories - definite, probable or unidentified, according to the confidence with which identification could be made (Mills et al. 1996). Only search phase pulses were analysed.

Definitions and abbreviations

F<sub>c</sub> = characteristic frequency (see Pennay et al. 2004 for parameter description)

DEF = definite

PROB = probable

UN = unidentified calls that could not be distinguished (species complexes) or that belonged to species that were not from the Mormopterus genus)

Data filtering steps

1. Exclude calls with < 3 pulses
2. Exclude calls where F<sub>c</sub> < 20 kHz or F<sub>c</sub> > 38 kHz
KEY:

1. Call sequence contains at least two flat-shaped pulses (and may include a number of curved/sloping pulses)  

2. >32 kHz

3. >10 flat pulses

4. Alternation (more than 1 in 3 pulses alternating or at least one alternation combined with characteristic downward sloping tails)  

4* No alternation (or less than 1 in 3 pulses alternating)  

M. norfolkensis DEF

3* 3-10 flat pulses

5. Alternation (more than half pulses or combined with characteristic downward sloping tails)  

5* Some alternation  

M. norfolkensis PROB

5# No alternation  

COMPLEX

2* <32 kHz

6. Upper Hunter (Nth Maitland)

7. <29kHz  

M. Sp.4 DEF

7* 29-30kHz  

M.Sp.2 DEF

7^ 30-32kHz

11. >10 flat pulses

12. Alternation (more than 1 in 3 pulses alternating or at least one alternation combined with characteristic downward sloping tails)  

12* No alternation  

M. Sp.2 DEF

11* 3-10 flat pulses

13. Alternation (more than one half of pulses or combined with characteristic downward sloping tails. If less than 4 pulses then must alternate)  

13* One to no alternation  

COMPLEX

1* Call has only sloping and/or curved-shaped pulses (only one flat-shaped pulse permitted)  

OTHER / COMPLEX
Appendix C: Candidate environmental variables

Seventeen environmental variables were selected for analysis of habitat use and were collected at both a site- and landscape-scale (see Table C1). Percent vegetation cover and percent urban land-use were calculated using the Lower Hunter and Central Coast Regional Biodiversity Conservation Strategy vegetation communities (House 2003), the ‘Vegetation of the Central Hunter Valley’ prepared by the Hunter-Central Rivers Catchment Management Authority (HCR CMA) (Peake 2006) and The Office of Environment and Heritage NSW regional land-use classification. Large forested areas on the edges of the Hunter Valley that were excluded from mapping by the HCR CMA (Peake 2006) were manually digitised from aerial photography. These analyses were undertaken using ArcGIS (version 9.0, ESRI, Redlands, CA, USA), a geographic information system.

Hourly climate data (rainfall, temperature, humidity and wind speed) were obtained from the nearest weather station (0.2 - 12 km; Bureau of Meteorology or private weather stations managed by coal mines) to each site. To determine whether temperature and humidity had an effect on bat activity during sampling, average nightly temperature and relative humidity were calculated from the hour of sunset to the hour of sunrise for each site. Additionally, average annual rainfall information for each site was obtained from the nearest meteorological station (0.2 - 12 km; Bureau of Meteorology) that had collected data for the previous 20 years as a minimum. Season was included as a variable as higher activity levels could occur during the weaning season (February – April) when young become volant and form part of the active population compared to the maternity season (November – January).

An index of net primary productivity (NPP) was developed to account for the possibility that bats move in response to variations (both spatial and temporal) in insect abundance. The NPP was calculated by averaging the plant productivity index values (Kesteven et al.
2004) within each site buffer and then summing the average plant productivity index values for the month of sampling and one month before and after sampling.

The slope and elevation layers were derived from a 25 m Digital Elevation Model (DEM) which was interpolated (drainage enforced) using 10 m contours and drainage lines / regions from 1:25,000 topographic maps. The landform feature layer was created for the study area following a landform classification system approach (Weiss 2001) as applied by Tagil and Jenness (2008) which combines a topographic position index (TPI) and slope. We applied broader classifications (0.5 SD) than the 1 SD specified by Weiss (2001) to allow the gentle, rolling features along the Hunter Valley to be included, whilst still identifying the steeper surrounding mountain range features.

Table C1: Description and source of candidate variables considered in the model building process

<table>
<thead>
<tr>
<th>Candidate Variable</th>
<th>Scale</th>
<th>Definition</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>LType</td>
<td>Landscape</td>
<td>Landscape type – original categorical stratification unit – cleared, semi-cleared, forested, urban</td>
<td>GIS analysis – vegetation and land-use datasets</td>
</tr>
<tr>
<td>LEType</td>
<td>Site</td>
<td>Habitat element type – original categorical stratification unit – open, forest remnant, paddock tree, riparian</td>
<td>GIS analysis – vegetation and land-use datasets</td>
</tr>
<tr>
<td>NearHBTlog</td>
<td>Site</td>
<td>Distance to the nearest hollow-bearing tree (m) – log(x+1) transformed</td>
<td>Site field data</td>
</tr>
<tr>
<td>CanHeightLow</td>
<td>Site</td>
<td>Canopy height binary category. Canopy &lt;15m = 1; &gt;15m = 0</td>
<td>Site field data</td>
</tr>
<tr>
<td>RipPres</td>
<td>Landscape (0.25, 1, 5, 10km buffers)</td>
<td>Binary riparian habitat category (&gt;5% riparian land-use = 1; &lt;5% riparian land-use = 0) within buffer of sample site. Major riparian habitats only</td>
<td>Land-use dataset (NSW OEH)</td>
</tr>
<tr>
<td>NearRipLUlog</td>
<td>Landscape</td>
<td>Distance to the nearest riparian land-use (m) log(x+1) transformed. Major riparian habitats only.</td>
<td>Land-use dataset (NSW OEH)</td>
</tr>
<tr>
<td>UrbPres</td>
<td>Landscape (0.25, 1, 5, 10km buffers)</td>
<td>Binary urban land-use category (&gt;10% urban land-use = 1; &lt;10% urban land-use = 0) within buffer of sample site</td>
<td>Land-use dataset (NSW OEH)</td>
</tr>
<tr>
<td>Candidate Variable</td>
<td>Scale</td>
<td>Definition</td>
<td>Source</td>
</tr>
<tr>
<td>--------------------</td>
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</tr>
<tr>
<td>Veg</td>
<td>Landscape (0.25, 1, 5, 10km buffers)</td>
<td>Area of vegetation within buffer of sample site (ha)</td>
<td>Vegetation dataset (adapted from House 2003, Peake 2006)</td>
</tr>
<tr>
<td>NPP</td>
<td>Landscape (0.25, 1, 5, 10km buffers)</td>
<td>Plant Productivity Index - Sum of the average of all cells within buffer of sample site for the month prior, month of and month after sampling</td>
<td>AGO plant productivity dataset (Kesteven et al. 2004)</td>
</tr>
<tr>
<td>Season</td>
<td>Other</td>
<td>Season of sampling: maternity (Nov-Jan); weaning (Feb-April);</td>
<td>Field data</td>
</tr>
<tr>
<td>DetVol</td>
<td>Site</td>
<td>Detector volume</td>
<td>Site field data calculated using AnaVolume (Corben 2009)</td>
</tr>
<tr>
<td>AvgAnnRain</td>
<td>Site</td>
<td>Average annual rainfall (ml)</td>
<td>BOM</td>
</tr>
<tr>
<td>AvgNightlyTemp</td>
<td>Site</td>
<td>Average nightly temperature (°C)</td>
<td>Nearest weather station (BOM or private)</td>
</tr>
<tr>
<td>AvgNightlyRH</td>
<td>Site</td>
<td>Average nightly relative humidity (%)</td>
<td>Nearest weather station (BOM or private)</td>
</tr>
<tr>
<td>Elevlog</td>
<td>Site</td>
<td>Elevation (m) log(x+1) transformed</td>
<td>GIS analysis - DEM</td>
</tr>
<tr>
<td>Slopelog</td>
<td>Site</td>
<td>Slope (°) log(x+1) transformed</td>
<td>GIS analysis - DEM</td>
</tr>
</tbody>
</table>

### Appendix D: Post-hoc model results

Table D1: Summary from *post-hoc* generalised linear mixed model analysis of landscape variables from 98 sites for all *Mormopterus* species

<table>
<thead>
<tr>
<th>Model</th>
<th>Factor(s)</th>
<th>Estimate ± SE</th>
<th>ΔAICc</th>
<th>W&lt;sub&gt;i&lt;/sub&gt;</th>
<th>AUC original dataset</th>
<th>AUC validation dataset</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Mormopterus norfolkensis</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MnModel A</td>
<td>Intercept</td>
<td>4.548 ± 1.878</td>
<td>0.00</td>
<td>0.14</td>
<td>0.90</td>
<td>0.58</td>
</tr>
<tr>
<td></td>
<td>Veg_1km</td>
<td>-0.006 ± 0.004</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td></td>
<td>Elevlog</td>
<td>-2.284 ± 0.957</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>UrbPres_10km</td>
<td>-2.331 ± 1.096</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MnModel B</td>
<td>Intercept</td>
<td>5.577 ± 1.943</td>
<td>0.94</td>
<td>0.08</td>
<td>0.91</td>
<td>0.57</td>
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<tr>
<td></td>
<td>NearRipLUlog</td>
<td>-0.448 ± 0.286</td>
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<tr>
<td></td>
<td>Elevlog</td>
<td>-2.513 ± 0.969</td>
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<tr>
<td></td>
<td>UrbPres_10km</td>
<td>-2.793 ± 1.092</td>
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<tr>
<td>MnModel C</td>
<td>Intercept</td>
<td>5.014 ± 1.907</td>
<td>1.00</td>
<td>0.08</td>
<td>0.91</td>
<td>0.60</td>
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<tr>
<td></td>
<td>Elevlog</td>
<td>-2.825 ± 0.955</td>
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<td></td>
<td>UrbPres_10km</td>
<td>-2.952 ± 1.098</td>
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<tr>
<td>MnModel D</td>
<td>Intercept</td>
<td>5.627 ± 2.013</td>
<td>1.42</td>
<td>0.07</td>
<td>0.91</td>
<td>0.58</td>
</tr>
<tr>
<td></td>
<td>LForm</td>
<td>-0.272 ± 0.201</td>
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<tr>
<td></td>
<td>Elevlog</td>
<td>-2.442 ± 1.017</td>
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<td>UrbPres_10km</td>
<td>-2.717 ± 1.113</td>
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<tr>
<td>MnModel E</td>
<td>Intercept</td>
<td>4.104 ± 1.951</td>
<td>1.97</td>
<td>0.05</td>
<td>0.90</td>
<td>0.59</td>
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<tr>
<td></td>
<td>Sloleplog</td>
<td>-0.662 ± 0.571</td>
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<td></td>
<td>Elevlog</td>
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<td>UrbPres_10km</td>
<td>-2.675 ± 1.065</td>
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<tr>
<td>MnModel F</td>
<td>Intercept</td>
<td>1.738 ± 1.048</td>
<td>1.74</td>
<td>0.06</td>
<td>0.87</td>
<td>0.51</td>
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<tr>
<td></td>
<td>Veg_1km</td>
<td>-0.008 ± 0.003</td>
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<tr>
<td></td>
<td>LForm</td>
<td>-0.338 ± 0.195</td>
<td></td>
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</tr>
<tr>
<td>Model</td>
<td>Factor(s)</td>
<td>Estimate ± SE</td>
<td>△AICc</td>
<td>W</td>
<td>AUC original dataset</td>
<td>AUC validation dataset</td>
</tr>
<tr>
<td>------------------------</td>
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NC – not calculated. The landscape cell ID was added as a random effect term in the model. Difference in Akaike information criterion adjusted for small sample size (ΔAICc) is compared to the best fitting model for each species. Model weights (W) are calculated from the AICc values of the 95% confidence set of models (29 models for M. norfolkensis, five for M. species 2 and 26 for M. species 4). Area under the curve (AUC) values were calculated for the receiver operating characteristic (ROC) based on both the training dataset (presence/absence records collected for this study) and for the testing dataset (regional presence-only database records with random points used as absences).
Chapter 3

Observations on the roost characteristics of the East-coast Freetailed Bat *Mormopterus norfolkensis* in two different regions of New South Wales

Anna McConville and Bradley Law

Australian Zoologist
Chapter 3 Observations on the roost characteristics of the East-coast Free-tailed Bat *Mormopterus norfolkensis* in two different regions of New South Wales

3.1 Abstract

The protection of day roosts is critical to the conservation of threatened insectivorous bat species. However, little is known about the roosting ecology of many species and this is particularly the case for Australian hollow-roosting species, such as East-coast Free-tailed Bat *Mormopterus norfolkensis*. We undertook targeted surveys to capture *M. norfolkensis* and investigate the factors that influence roost selection. However, we were only able to capture and radio-track six individuals in two different regions of New South Wales (three in the Hunter Valley and three at Urbenville). We found that *M. norfolkensis* roosted in tree hollows in a range of tree species, including Grey Box *Eucalyptus moluccana* and Spotted Gum *Corymbia maculata* in the Hunter Valley and Flooded Gum *E. grandis* and red gum *E. amplifolia / tereticornis* in Urbenville. Additionally, a telegraph pole was used as a roost by a small colony of eight bats in Urbenville. As we experienced very low trap success and were only able to track bats for a brief period, roost preference is yet to be thoroughly investigated.

3.2 Introduction

There is a paucity of information available on the ecology of Australian insectivorous bat species. The least well known of these tend to be high-flying, hollow-roosting species that are not often captured and many are listed as threatened under state and federal
legislation. Bats depend on day roosts to fulfil a number of life-cycle functions and a lack of roosts may limit their distribution and the size of populations (Kunz and Lumsden 2006, Churchill 2008). At present, conservation management strategies can only be directed towards a select few hollow-roosting species that have been adequately studied (Lunney et al. 1988, Law and Anderson 2000, Lumsden et al. 2002a, Webala et al. 2010, Threlfall et al. 2013a). Furthermore, compensatory habitat strategies require roost selection information to ensure that offsets contain suitable roosting habitat for threatened bat species.

One hollow-roosting insectivorous bat of which little is known is the East-coast Free-tailed Bat *Mormopterus norfolkensis* (Figure 3-1). It occurs along the east coast of Australia and is listed as Vulnerable under the New South Wales (NSW) Threatened Species Conservation Act 1995 and as Vulnerable C1 under the International Union for Conservation of Nature (IUCN) red list (IUCN 2009). While the species has been anecdotally recorded from natural and artificial roosts when these habitats come into conflict with humans (i.e. during tree felling operations and in buildings), there is no empirical evidence to quantify what type of roosts are most suitable.
We conducted targeted surveys to capture *M. norfolkensis* for radio-tracking studies to investigate what factors influence roost selection. However, due to the small number of individuals captured, we revised our aims to simply document and discuss the characteristics of roosts that were used by *M. norfolkensis*.

### 3.3 Methods

#### 3.3.1 Pokolbin - Lovedale study area, Hunter Valley

Pokolbin and Lovedale are located in the Hunter Valley, NSW, 45 km inland from the port of Newcastle on the east coast of Australia (Figure 3-2). The predominant land-uses in this rural locality are viticulture and cattle grazing, with some tourist facilities also occurring. Much of the surrounding area has been cleared for agriculture, with remnant vegetation existing mainly as scattered paddock trees and narrow linear patches along roads and creek-lines. Dominant tree species are Spotted Gum *Corymbia maculata,*
Narrow-leaved Ironbark *Eucalyptus crebra* and Red Ironbark *E. fibrosa*, with Grey Box *E. moluccana* also occurring. Large forest reserves exist approximately 6 km to the west in the sandstone escarpment of the Broken Back Ranges (Pokolbin State Forest) and to the east in more disturbed low elevation remnants (Werakata National Park). While the floodplains of the Hunter River consist of relatively rich alluvial soils, the study area occurs on the fringe of the floodplain more than 10 km away from the Hunter River and so it is also influenced by the poor soil quality associated with the sandstone escarpments.

Smaller areas of freshwater wetland, Swamp Oak *Casuarina glauca* and River Oak *C. cunninghamiana* forests once occurred in riparian habitats and some disturbed remnants remain. Additionally, many large farm dams occur in low-lying areas across the study area to irrigate vineyards and to provide water to stock. In the Hunter region, *M. norfolkensis* is distributed much farther west than anywhere else in NSW (The Office of Environment and Heritage 2011), as the mountains of the Great Dividing Range are relatively low and gentle at the broad Hunter Valley (Peake 2006). The Pokolbin - Lovedale study area is located in the centre of the east-west distribution of *M. norfolkensis* in the Hunter region.

We selected Pokolbin and Lovedale as our primary study area as it was conducive to radio-tracking (it had a good road network, line of site and reasonable private property access) and most importantly high (but patchy) levels of *M. norfolkensis* activity (its distinctive echolocation call identified from bat call recordings collected during a systematic study; Chapter 2). This rural landscape was vastly different to forested reserves and large remnant patches that have been traditionally targeted for trapping of insectivorous bats, which have been largely unsuccessful in capturing *M. norfolkensis* (e.g. NSW National Parks and Wildlife Service 1998).
3.3.2 Urbenville study area

The Urbenville study area was located in far northern NSW, close to the Queensland border and approximately 93 km inland (Figure 3-2). The study area is part of a long term research project investigating the habitat value of paddock trees and eucalypt plantations in agricultural areas of NSW, where high *M. norfolkensis* activity had been recorded (Law et al. 2000) and one male had been radio-tracked (B. Law unpublished data). Urbenville is located at the western extent of *M. norfolkensis* distribution in northern NSW (The Office of Environment and Heritage 2011). Urbenville is a small rural community where the predominant land-uses are agriculture on the fertile alluvial flats and forestry on the surrounding volcanic ranges. The agricultural areas were characterised by undulating hills with small flats along creeks. The remnant vegetation consisted mainly of large old paddock trees such as Broad-leaved Apple *Angophora subvelutina*, Small-fruitied Grey Gum *E. propinqua*, Pink Bloodwood *C. intermedia* and Tallowwood *E. microcorys*. Grey Ironbark *E. sideroploia* dominated drier topography such as ridge-tops and Cabbage Gum.
E. amplifolia occurred in riparian zones. Yabbra State Forest occurs in the high relief eastern portion of the study area and consists of dry sclerophyll forests on the elevated slopes grading to wet sclerophyll forest and rainforest in the gullies. Since 2000, some of the agricultural areas adjoining existing forest areas have been converted to eucalypt plantation comprising Blackbutt E. pilularis, Dunn’s White Gum E. dunnii, and C. maculata.

Investigative trapping was undertaken at two other locations in NSW, Coolongolook and Paterson. However, M. norfolkensis was not captured and these sites are not discussed here further.

### 3.3.3 Trapping

We undertook a targeted trapping survey in November - December 2008 using harp traps, mist nets and trip lines in the Hunter Valley to capture M. norfolkensis for radio-tracking. As the morphological and anecdotal evidence suggests that M. norfolkensis is an open-adapted species (Rhodes 2002, Churchill 2008, Hoye et al. 2008), we located trapping sites in forests with an open structure or in semi-cleared agricultural areas. We placed traps along tracks in forested reserves (Werakata National Park) and we also set harp traps near paddock trees or groups of paddock trees in private rural properties. Some of these harp traps (three traps for a total survey effort of 18 harp trap nights) were hoisted up under overhanging branches of trees to a height of approximately 6 m high to target higher-flying bats both along forest tracks and near paddock trees. We set trip-lines and monofilament mist nets at small dams to capture bats as they flew low to drink. We also set mist nets on 4 m poles near paddock trees or forest edges where we thought bats might be regularly travelling.
At Urbenville, trapping was undertaken in November 2009 as part of a larger study on wildlife succession in eucalypt plantations (B. Law unpublished data). Harp traps were positioned to take advantage of flyways along tracks in the 11 year old eucalypt plantation, especially along elevated narrow ridges and near water-bodies to target *M. norfolkensis* and Greater Broad-nosed Bat *Scoteanax rueppellii* for radio-tracking studies (B. Law unpublished data). One harp trap was hoisted up under an overhanging branch to approximately 6 m height to target higher-flying bats such as *M. norfolkensis* for two nights.

### 3.3.4 Radio-tracking

Transmitters (Titley Scientific, LT4-337, ~0.4 g, single stage with 30 cm whip antennae) were positioned mid-dorsally below the scapulæ and fixed in place using surgical glue (VetBond, 3M, St Paul, MN; Figure 3-3). Bats were released after transmitter attachment before dawn or within two hours of dusk if they were held during the day. Searches for radio-tagged bats were conducted during the day by vehicle and by foot along accessible tracks within the study area with regular targeted searches conducted at high vantage points. Once a signal was obtained, directional antennae were used to track bats to their roosts by homing in on the signal (White and Garrot 1990) until the roost or roost tree was identified. We recorded the following at each roost: type of roost (e.g. tree hollow, dead limb, under bark); roost height; entrance diameter of roost; tree species; tree height; diameter at breast height over bark (DBH); senescence level (of increasing senescence categories from 1 - 8, following Gibbons et al. 2000); projected foliage cover of the canopy (PFC); distance to nearest tree; distance to nearest hollow-bearing tree; and distance to water.
Roosts were stag-watched, where possible, for 30 mins prior and one hour after dusk, with bats counted and the roost location confirmed. A bat detector (Anabat SD1, Titley Electronics, Balina, Australia) was carried during stag-watching to record bat echolocation calls and to assist in identifying the bat species present. Stag-watching was not conducted within the wet sclerophyll forest at Urbenville as the height of the emergent roost trees and the dense rainforest canopy meant that it was highly unlikely that we would be able to observe bats exiting.

Searches were also undertaken for bats at night using directional antennae from vehicles to confirm that bats had not lost transmitters (for those roosts that were not able to be stag-watched) and to determine where they were foraging. Bats were followed via vehicle as they moved about the landscape, with distances estimated from signal strength,
calibrated from occasions when the bats crossed our path. Due to the small overall number of roosts and data points recorded at night, data were not statistically analysed.

3.4 Results

Low capture rates were found for *M. norfolkensis* in the Hunter Valley with a total of 240 harp trap nights, 18.75 mist net hours and 3.9 trip-lining hours undertaken during the six weeks of study. This resulted in a trap success of 1.25 *M. norfolkensis* individuals per 100 harp trap nights. Survey effort was not even or stratified across the study area, with successful locations subjected to more trapping effort. For example, we repeatedly trapped the site where the first *M. norfolkensis* (M2) was captured to try to capture more individuals, with a total of 21 harp trap nights undertaken at this one site. This approach was successful in capturing another *M. norfolkensis* individual (M4) towards the end of the Hunter Valley study period, resulting in a higher trap success rate of 9.5 *M. norfolkensis* per 100 harp trap nights at this one site. At Urbenville, a total of 30 harp trap nights, four mist net hours and three trip-lining hours were undertaken in 2009, resulting in a capture rate of 13.3 *M. norfolkensis* per 100 harp trap nights. It should be noted that considerably more survey effort has been undertaken prior to this study within the Urbenville study area since 1997 with only one *M. norfolkensis* captured (B. Law, unpublished data). Mist nets and trip-lines were unsuccessful at capturing *M. norfolkensis* in both study areas. Most transmitters were removed by bats quickly during the study, with most bats tracked for only one day (Table 3-1).

3.4.1 Hunter Valley

A total of three *M. norfolkensis*, two males and one non-breeding female, were tracked to three different roosts in the Pokolbin - Lovedale study area, November - December 2008. Two bats (M2 and M4) were captured in a harp trap placed next to a linear roadside
remnant and one (M3) was captured in a small patch of paddock trees in a grazing pasture within 640 m of the other successful trap (Figure 3-4). One roost was identified for each of the bats, with the two males tracked for two days each and the female for only one day (Table 3-1). Two _E. moluccana_ and one _C. maculata_ were used as roosts in the Hunter Valley with small colony sizes of 1 - 2 bats recorded (Table 3-1). Bats moved less than 2 km from capture site to roost and often the bats roosted close to where they were trapped (Figure 3-4).

**Figure 3-4: Hunter Valley roost and trap locations**

Hunter Valley _Mormopterus norfolkensis_ roost (yellow circle) and successful trap (red triangle) locations. A 10 km radius surrounding the trap and roost locations (yellow line) is shown to illustrate the maximum distance likely to be travelled by radio-tracked bats. Mapped using Google earth (version 6.2.2.6613).
One male (M2) was tracked over three nights (including the night of release) for a total of 1.5 hours between 2030 - 2400 h and during this time it stayed relatively close (maximum distance of 1.5 km) to its roost in a *E. moluccana* (M2.1; Figure 3-5; Table 3-1). M2 roosted in a small dead branch with a small hollow in the end (approximately 2 cm diameter) and the tree was located < 2 m from the edge of a road in a linear roadside remnant (Figure 3-5). Adjacent to the roost tree was a large, unfenced block with regenerating eucalypts and shrubs, with holiday cabins with a mown understorey and scattered trees occurring on the opposite side of the road to the roost. M2 appeared to travel along a small power line easement and also spent time in an open woodland remnant with very little understorey and the low-density holiday cabins. Large dams and remnant riparian vegetation were also present within the general area surrounding the roost.
Another male (M3) was tracked over two nights (including the night of release) for a total of 2.25 hours between 2130 - 2330 h, to a maximum distance of 2 km from the roost. This individual appeared to traverse open paddocks, used for cattle grazing, quite quickly after release and then moved back and forth over a linear-shaped area approximately 120 ha in size that contained remnant riparian vegetation and paddock trees. M3 roosted for two consecutive days in a large *C. maculata* paddock tree on a rural residential property, approximately 70 m from a house (M3.1; Figure 3-6; Table 3-1). The roost entrance was an elongated fissure on the trunk, approximately 30 cm long and up to 15 cm wide, with a westerly aspect (Figure 3-6; Table 3-1). The general area surrounding the roost had little understorey (grazing and mowing), with patches of young eucalypts, scattered old paddock trees and some ornamental shrubs.
The non-breeding female bat (M4) was tracked on the night of release from 2100 - 2330 h and from 0330 - 0450 h. Early in the night, after initially travelling approximately 2 km, M4 spent over one hour moving around a remnant open woodland patch approximately 11 ha in size that was located 0.5 km from the capture site. Before dawn the next morning, M4 was located 1.3 km from the capture site and then was recorded moving progressively closer to the roost during the next hour. M4 was tracked until it was found to be stationary within the roost (M4.1; Figure 3-7; Table 3-1) at 0450 h, 39 minutes before civil twilight.
However, on return later that morning the signal was not able to be relocated, despite searching the surrounding road network thoroughly. The roost (M4.1) was stag-watched with a bat detector on dusk and two *M. norfolkensis* (identified by call) were observed to exit. We concluded that the transmitter was likely to have ceased operating while the bat was inside the roost.

Figure 3-7: *Mormopterus norfolkensis* roost tree (M4.1), Grey Box *Eucalyptus moluccana*, Hunter Valley, NSW

Photo: A. McConville
3.4.2 Urbenville

Three *M. norfolkensis* were radio-tracked at Urbenville during November 2009 (one non-breeding female, one lactating female and one pregnant female; Table 3-1). Another lactating female was captured towards the end of the study, but it was not radio-tracked. The bats were all captured in a single harp trap which was placed along a ridge-top track in the young eucalypt plantation (Figure 3-8). The non-breeding bat (MU2) was tracked for five consecutive days with two roosts located and the other bats were tracked for one day only before their transmitters were removed. A total of four roost trees were identified (Figure 3-8), two of these being maternity roosts. However, only one of the roost entrances was confirmed.
Urbenville Mormopterus norfolkensis roost (yellow circle) and successful trap (red triangle) locations. A 10 km radius of the trap and roost locations is shown (yellow line) to illustrate the maximum distance likely to be travelled by radio-tracked bats. Mapped using Google earth (version 6.2.2.6613).

The non-breeding female (MU2) roosted in two different trees in a rainforest gully with tall (40 - 45 m) emergent E. grandis for a total of five days (Figure 3-8; Figure 3-9). Whilst the actual roost locations in the trees were not confirmed, they were estimated to be high based on signal strength and triangulation. Whilst the PFC of the canopy in the rainforest gully was estimated to be 75 %, the PFC of the emergent layer where the bats were thought to be roosting was just 20 % being more similar to a woodland structure when above the dense rainforest canopy.
Urbenville rainforest gully in which a non-breeding female *Mormopterus norfolkensis* (MU2) roosted in two different *Eucalyptus grandis* Flooded Gum for a total of five days, near Urbenville, NSW. Photo: B. Law

The pregnant female (MU1) roosted in a telegraph pole located in full sun (MU1.1; Figure 3-8) in a small, partially cleared paddock near a creek. The colony, of eight individuals, exited the telegraph pole from under the metal cap between 1950 - 2000 h, with MU1 emerging at 2000 h which was 43 mins after sunset and 18 mins after civil twilight. However, it is unknown if the bats were roosting directly under the hot metal cap or whether they were within a more protected central cavity. The maximum temperature recorded at the nearest weather station was 25.1 °C (Tabulam station, Bureau of Meteorology) and it is likely to have been much hotter under the exposed metal cap compared to the surrounding available hollows.
The lactating female (MU3) roosted in a patch of remnant red gum (either Forest Red Gum *E. tereticornis* or *E. amplifolia*) paddock trees (Figure 3-8; Figure 3-10). However, the exact roost location was unable to be determined during stag-watching as the transmitter appeared to stop functioning prior to bats exiting. We are reasonably confident that the transmitter mal-functioned as searches for MU3 that same night and subsequent days and nights failed to locate any signal.

**Figure 3-10: Urbenville red gum paddock trees**

Patch of remnant red gum paddock trees (either Forest Red Gum *E. tereticornis* or Cabbage Gum *E. amplifolia*) where a lactating female *Mormopterus norfolkensis* (MU3) roosted near Urbenville, NSW. Photo: B. Law

At Urbenville, searches were made for tracked bats from 2100 - 2400 h each night. Signals were more intermittent than during the Hunter Valley component of the study,
probably due to poor signal reception resulting from undulating terrain and relatively dense eucalypt plantations at Urbenville, compared with the open pasture in the Hunter Valley. However, we did record the non-breeding female (MU2) from within the plantation forest each night and activity appeared to be focused along a small alluvial flat approximately 500 m from the trap site. We recorded this bat in this area for 1.25 hours on one night and for 20 mins on the following night before we left to search for other bats. This area had scattered remnant trees along an ephemeral creek-line and an open alluvial flat which was bordered upslope by eucalypt plantation. The maximum distance that *M. norfolkensis* was recorded travelling at Urbenville was 5 km from capture site to the rainforest gully roost (Table 3-1; Figure 3-8).
### Table 3-1: Mormopterus norfolkensis roost details

DBH = diameter at breast height; HBT = hollow-bearing tree; H = tree hollow; A = Artificial; NA = not observed. Senescence category is a 1 – 8 scale of increasing senescence following Gibbons et al. 2000

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<th>Bat ID</th>
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<th>Tree Species</th>
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<th>Senescence category</th>
<th>Roost tree Height (m)</th>
<th>Entrance diameter of roost (cm)</th>
<th>Distance to nearest tree (m)</th>
<th>Distance to nearest HBT (m)</th>
<th>Distance to water (m)</th>
<th>PFC</th>
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<td>Grey Box</td>
<td>66</td>
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<td>Roost Type</td>
<td>Senescence category</td>
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<td>Roost Height (m)</td>
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<td>Distance to nearest HBT (m)</td>
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<td>Colony Count</td>
<td>Distance trap to roost (m)</td>
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3.5 Discussion

This is the first published study to investigate roost use by *M. norfolkensis* and whilst we planned to have a greater sample size, we were only able to capture seven individuals and track six of them for a very short period. Our study confirms the difficulty of capturing rare open-adapted insectivorous bats, even in areas where the species has been regularly recorded using bat detectors. As such, *M. norfolkensis* roost preference remains to be revealed and future studies need to carefully consider ways to increase sample sizes.

Despite these sample size limitations we did make some new discoveries, recording roosts in some unexpected locations. The rainforest gully that a non-breeding female roosted in at Urbenville was unforeseen. We anticipated that the dense vegetation within the gully would have precluded open-adapted bat species from using the area. However, it is possible that *M. norfolkensis* did not fly below the rainforest canopy, but used the airspace above where a more open structure amongst the emergent trees occurred. Additionally, whilst *M. norfolkensis* has been reported roosting under telegraph pole caps previously (Churchill 2008, Hoye et al. 2008), it was surprising to find this at Urbenville. Despite the agricultural and forestry history of the Urbenville study area there were many hollow-bearing trees persisting as paddock trees or embedded within the plantation (see Law et al. 2000) and indeed there were 5 - 10 large hollow-bearing trees within 50 m of the telegraph pole. It is possible that the exposed telegraph pole offered some thermoregulatory benefit to roosting bats compared with the nearby hollows and further research into use of these artificial structures would be valuable. Extensive dusk stag-watches of paddock trees over a 12 year period at Urbenville, aided by bat detectors, have failed to observe any *M. norfolkensis* roosts, although it is possible that roosts of individual bats were missed (Law et al. 2000; B. Law unpublished data). When these
observations are considered with the results from this radio-tracking study, it appears that *M. norfolkensis* roosts in small colonies.

We also recorded *M. norfolkensis* travelling relatively short distances (maximum 2 km) in the Hunter Valley for an open-adapted species. Bats were tracked travelling greater distances (maximum 5 km) in Urbenville and this is consistent with previous radio-tracking at Urbenville in May 2002, where a female *M. norfolkensis* was tracked 6 km from its roost in an isolated *E. tereticornis* to foraging areas around a cemetery on the outskirts of town (B. Law, unpublished data). However, this was still less than what we expected based on other Australian *Mormopterus* species. For example, the South-eastern free-tailed bat *Mormopterus* species 4 has been reported travelling 12 km from a roost to forage (Lumsden et al. 2008). However, this is not to say that *M. norfolkensis* does not travel large distances on occasion, or regularly in other regions.

Finally, in the Hunter Valley, our study illustrates that in landscapes where much of the native vegetation has been removed, roadside reserves can be important roosting habitat for threatened insectivorous bat species.

### 3.6 Acknowledgements

This project was undertaken during a PhD candidature (AM) and we would firstly like to thank Michael Mahony for his supervision and advice. Two anonymous referees also provided helpful comments. Kearsley Rural Fire Service and in particular, Les and Jane Goldie generously provided access to their facilities during trapping in the Hunter Valley. We would also like to thank the very patient volunteers, in particular, Jordan Crabbe that assisted in the Hunter Valley and the NSW Primary Industries Forest Science Centre staff, especially Mark Chidel and Alison Towerton, for assisting with radio-tracking in Urbenville.
Additionally, we would like to thank the property owners and managers that provided access to their land to set traps and locate roosts. Funding for this PhD project was sourced from: The Tom Farrell Institute for the Environment; Donaldson Conservation Trust; Hunter-Central Rivers Catchment Management Authority; Royal Zoological Society of NSW (Ethel Mary Read Award); Wambo Coal Pty Ltd; and Australian Geographic Society. This project was conducted under scientific licence and animal ethics approvals, under the *Australian Code of Practice for the Care and Use of Animals for Scientific Purposes, 7th edition, 2004*. 
Mangroves as maternity roosts for a colony of the rare east-coast free-tailed bat (*Mormopterus norfolkensis*) in south-eastern Australia

Anna McConville, Bradley Law and Michael Mahony

Wildlife Research
Chapter 4 Mangroves as maternity roosts for a colony of the rare east-coast free-tailed bat (Mormopterus norfolkensis) in south-eastern Australia

4.1 Abstract

Context
Maternity roosts of insectivorous bats (where females raise young) are critical to the conservation of threatened species as roost quality can influence reproductive success. Additionally, threatened species may have specialised requirements or unusual behaviour, which may be overlooked without targeted investigation.

Aims
To explore which factors influence Mormopterus norfolkensis roost selection by comparing day roosts, identified via radio-tracking, with environmental variables collected at tree, patch and landscape scales.

Methods
We collected a range of variables describing maternity roosts and surrounding patches, including internal measurements of hollows and microclimate. Additionally, we derived landscape-scale variables using a geographic information system. We then explored which variables best explained roost occurrence by comparing roosts to available trees using logistic regression.

Key results
Nineteen lactating females and two male M. norfolkensis were tracked to 40 roost trees, mostly grey mangrove Avicennia marina subsp. australasica. Lactating females were found to be faithful to two patches of mangrove forest close to where they were captured,
regularly switched roosts and roosted in hollows singularly or in small groups. Our models indicated that the attributes of mangrove patches, especially a high proportion of hollows, were more important to lactating females than roost tree or landscape characteristics. Additionally, whilst the microclimate of roost hollows was not significantly different from ambient mangrove conditions, the mangrove forest was slightly more stable and had higher humidity compared to other nearby habitats.

Conclusions

Contrary to predictions, maternity roosting group sizes were relatively small, indicating that bats were not deriving thermoregulatory benefits from communal living. However, we suggest that lactating females may benefit from the operation of a fission-fusion society among the colony as a whole. Additionally, the mature mangrove forest could offer unique roosting opportunities for bats as they support high densities of hollow-bearing trees, a stable microclimate and potentially low abundances of predators and competitors.

Implications

This is one of few international bat-mangrove studies and it illustrates that threatened species can behave unexpectedly and may be overlooked in conservation strategies that are based largely on anecdotal observations. We encourage further research into the value of mangrove forests to terrestrial fauna globally.

4.2 Introduction

Insectivorous bats depend on day roosts to fulfil many aspects of their lifecycle, including shelter from weather, protection from predators, sites for raising young and conditions to meet thermoregulatory requirements (Kunz and Lumsden 2006). A lack of suitable roosts may limit the distribution and population sizes of bats. Additionally, the quality of maternity roosts (where females raise young) may affect reproductive success through influences on juvenile survival, growth and development (Kunz and Lumsden 2006). It is clear that
the protection of habitat features that are required to fulfil so many important lifecycle functions is critical to the conservation of threatened bats and studies that investigate these requirements are valuable (Hurst and Lacki 1999, Law and Chidel 2004, Russo et al. 2004, Sedgeley 2006, Popa-Lisseanu et al. 2008, Timpone et al. 2010). However, in Australia, rigorous scientific data are lacking for many threatened insectivorous bat species and as such conservation management strategies often rely on anecdotal descriptions of habitat (see Armstrong 2011, Milne and Pavey 2011, Pennay et al. 2011 for conservation status reviews).

Insectivorous bats have a high surface area to volume ratio resulting from being small and having thin wing membranes and as such they have high energetic demands for thermoregulation despite being heterothermic. A number of strategies are thought to be employed to minimise these costs such as the selection of roosts with suitable microclimates (Kerth et al. 2001, Sedgeley 2001, Ruczynski and Bogdanowicz 2005, Willis and Brigham 2005) and communal roosting (Sedgeley 2001, Willis and Brigham 2007). Pregnant and lactating females are subject to even greater energetic demands associated with reproduction as they produce milk and carry large dependent offspring (often > 30% of adult size at birth) through pregnancy and between day roosts until weaning (3 - 10 weeks; Richards 2008). As such, it is likely that breeding females face intense selection pressure to choose roosts with characteristics that are a compromise between those that minimise energy expenditure and those that maximise reproductive success. There is variation in the roost selection strategies employed by different species and these can vary seasonally (Kunz and Lumsden 2006). However, the reasons for this are poorly understood.

The benefits of sociality at roosts, such as co-operative breeding (Emlen 1991), increased foraging efficiency (Beauchamp 1999) and the detection and evasion of predators
are constantly being traded off against the costs. These costs include direct competition for resources (Milinski and Parker 1991), increased chance of detection by predators (Vine 1973), greater susceptibility to the spread of parasites and diseases (Brown and Brown 1986) and difficulties associated with group decision-making and conflict resolution (Kerth et al. 2006). Maternity colonies of many hollow-roosting bat species have been proposed to operate as fission-fusion societies (eg Kerth and Konig 1999, O’Donnell 2000, Willis and Brigham 2004, Rhodes 2007, Popa-Lisseanu et al. 2008). In these societies, the colony consists of many small groups that are distributed among different day roosts (hereafter referred to as ‘roosting groups’). The day roosts are switched regularly over time, but individuals remain faithful to the patch, so that eventually, individuals mix among the colony as a whole (Kerth and Konig 1999, Willis et al. 2003, Willis et al. 2006, Popa-Lisseanu et al. 2008). In this way, fission-fusion societies are thought to gain the benefits of sociality, reduce parasite loads (Reckardt and Kerth 2007) and reduce predator detection, by switching roosts often (Lewis 1995).

One insectivorous bat that uses tree hollows as roosts and of which little is known is *Mormopterus norfolkensis* Gray, 1839 (east-coast free-tailed bat). It occurs on the east coast of Australia and is listed as vulnerable under the New South Wales (NSW) *Threatened Species Conservation Act 1995* and as vulnerable C1 under the International Union for Conservation of Nature (IUCN) red list (IUCN 2009). However, the species is difficult to capture (Chapter 3) and despite the conservation status of the species, roost selection by *M. norfolkensis* has yet to be rigorously explored. Anecdotal reports (Churchill 2008, Hoye et al. 2008) and limited data (Chapter 3) indicate that *M. norfolkensis* roosts in tree hollows, buildings, telegraph poles and nest-boxes.

The aim of this study was to explore which factors influence *M. norfolkensis* roost selection by comparing day roosts identified via radio-tracking with environmental
variables collected at tree, patch and landscape scales. A particular focus was given to maternity roosts as these are a high priority for conservation. On the basis of previous studies of insectivorous bats, we predicted that lactating females would roost: 1) communally (Kunz and Lumsden 2006); 2) in large trees (diameter and height; Law and Anderson 2000, Lumsden et al. 2002b, Webala et al. 2010); 3) in well insulated hollows that provide a warm, stable microclimate to minimise energy expenditure while lactating (Kerth et al. 2001, Sedgeley 2001, Willis and Brigham 2005); and 4) switch roosts regularly within patches to which they are faithful, as seen in fission-fusion social systems (Kerth and Konig 1999, Popa-Lisseanu et al. 2008).

4.3 Methods

4.3.1 Study area

The study was conducted in the Hunter Estuary, NSW, where the Hunter River enters the ocean at the Port of Newcastle (32°55’36"S 151°46’44"E; Figure 4-1). The area experiences a warm temperate climate (average monthly temperatures 8.4 – 25.6 °C) and average annual rainfall of 1134 mm (Bureau of Meteorolgy 2012). Bats were captured for radio-tracking in the western portion of what is now a 2600 ha landmass known as Kooragang Island. Areas surrounding Kooragang Island have been highly modified by a long history of agriculture, coal mining and port-associated industries. The non-industrial areas of Kooragang Island are dominated by mangroves and saltmarsh, with pasture and smaller areas of freshwater wetland also occurring. Hexham Swamp, a large and mostly freshwater wetland, occurs to the west of Kooragang Island. See Appendix A for further description.
4.3.2 Radio-tracking

We captured bats using harp traps (Austbat P/L, Victoria, Australia) that were mostly set in mangrove forests and attached radio-transmitters (≤ 5 % of the weight of individuals) to the dorsal surface of bats. Diurnal roosts were located each day for the battery life of the transmitter, or until the transmitter fell off. Additionally, roost trees were watched by observers from 30 min before dusk to 1 h after dusk. Roosting group size, time of tracked
bat exit, exit direction, hollow aspect and various hollow dimensions were recorded. See Appendix B for detailed radio-tracking methods.

4.3.3 Roost trees and plots

To investigate roost selection at a tree and patch scale, we collected a range of parameters from roost trees and also from a 10 m radius plot surrounding each roost tree (Appendix C). Measurements were collected from every woody stem within the plot and a stem was defined as a woody trunk, taller than breast height and > 3 cm diameter measured with a tape at breast height over bark (DBH). In this way trees with more than one trunk were treated as a separate stem so that stem density could be calculated to reflect vegetation clutter. The DBH, senescence category and whether hollows (≥ 1.5 cm diameter, assessed visually from the ground) were present were recorded for all stems within each plot. The number of stems and hollow-bearing stems (hereafter referred to as ‘hollow-bearing trees’) were later converted to densities per hectare. Additionally, a hollow-bearing tree abundance index (HBTAbundIndex) was calculated by dividing the number of hollow-bearing trees by the number of stems in a plot, where a value of 1 indicates that all stems contained hollows and values close to 0 indicate very few hollow-bearing trees in the plot.

To compare roost trees with others available to bats, we also measured the characteristics of available trees and surrounding 10 m radius plots following the same methodology as described for roosts above. We defined an ‘available tree’ as a tree (alive or dead) that had a hollow entrance ≥ 1.5 cm in diameter as this represented the type of roosts used by tracked bats. Available trees were selected from habitats within the maximum predicted nightly flight distance of roost trees (10 km radius; Chapter 6). Due to the large area of the 10 km radius, low abundance of available (hollow-bearing) trees and private property access issues, we did not randomly select sites. Instead, we used sites
previously selected for a related camera trapping study (A. McConville unpublished data 2012), where sampling was stratified by major vegetation type: Hunter Estuary mature mangrove forest; Kooragang Island mature mangrove forest; swamp oak forest patches; and dry sclerophyll open forest or woodland. When selecting sites, we chose examples of each of these vegetation types within the 10 km radius that ensured a spread of sites surrounding the roosts. At each of the sample sites we selected the nearest available tree to the camera trap location as the centre tree for plots. Two plots separated by 100 m were undertaken in patches of mature mangrove forest that were difficult to access. Vegetation patches which did not contain any hollow-bearing trees (such as young regrowth) were avoided as plots required an available (i.e. hollow-bearing) tree as a centre tree and as such, available plots were biased towards habitats that contained hollow-bearing trees, providing a best-case estimate of these resources within 10 km of roosts. Direct comparison with other studies should be undertaken with this, and the small size of hollow classification, in mind.

Additionally, we calculated landscape variables using ArcGIS (ESRI, Redlands, California, USA, version 9.3) in 500 m and 1 km buffers around all roost and random trees (Appendix C). The layers used were woody vegetation, freshwater wetland, mature mangrove forest and urban land-use. Layers were manually digitised within a 10 km buffer of all the roosts using aerial photography and were assigned a type on the basis of a combination of prior knowledge, site inspection and review of regional vegetation mapping and land-use datasets.

4.3.4 Hollow inspections

We characterised the internal characteristics of hollows using a small camera attached to a flexible cable (Seesnake, Rapid, USA) and also used this to calibrate visual estimates of mangrove hollows. As we used a ladder to access hollows, only hollows < 4 m in height
(half the height of the highest roost recorded) were examined. As such inspections of 12 roost hollows (50 % of all roost hollows identified) and 34 available hollows (42 % of all available hollows identified) were undertaken. Available hollows were those in the closest two hollow-bearing trees to nine randomly selected roost trees. We recorded hollow entrance dimensions, maximum internal dimensions, hollow depth, height and the presence of animals.

4.3.5 Microclimate

To compare temperature differences between mangroves and other habitats within 10 km of roosts, we used thermochron ibuttons (Maxim, San Jose, USA; range: -40 - 85 °C, accuracy: ± 1 °C) during 2009 / 2010. We then used universal serial bus (USB) data-loggers (range: 0 – 100% RH, -40 - 70 °C; accuracy: ± 3 % RH, ± 0.5 °C) during 2012, to compare temperature and relative humidity in mangrove roost hollows, nearest available hollow and ambient mangrove conditions. We also compared these temperature and relative humidity measurements with those taken from nearby weather stations (Bureau of Meteorology, 2012). We positioned ambient data-loggers in sheltered locations away from direct sunlight and fitted plastic conical hats over the data-loggers allowing 5 cm to protrude to minimise the effects of sunlight and rain, without substantially reducing airflow. We sampled each of the locations simultaneously for five days and calculated average hourly climatic conditions (e.g. 13 hr is the average of conditions between 12:01 – 13:00 hr) and the hourly rate of change by subtracting the conditions from the previous hour. Following Sedgeley (2001), we then further divided data into an average 24 hr period, average day (> 6:00 and < 20:00 hr) and average night (> 19:00 and < 7:00 hr) conditions. A total of three ibutton replicates and two USB data-logger replicates were measured.
4.3.6 Statistics

Analyses were undertaken for maternity roosts only. We used roosts as the experimental unit and as such, we have assumed that roosts were independent observations and were not biased by individual bat preferences. Heavily skewed variables were logarithmic (log(x + 1)) transformed prior to analysis and variables were converted to categories when they could not be transformed to a normal distribution (Appendix C). We calculated an index of roost re-use for each bat by dividing the total number of days that the bat was tracked, by the total number of roosts used by that particular bat (excluding any days that the bat could not be found). We separated the available tree plots into mangroves and other trees, as the mangroves had very different characteristics which resulted in bimodal distributions for many variables when they were pooled together. We used a paired-sample t-test to compare temperature between mangroves and other habitats using JMP (SAS Institute, version 9.0). All results are presented as mean ± standard error.

To explore the differences between roosts, available mangroves and available other habitats, we firstly undertook a Principle Components Analysis (PCA; see Appendix D). We then constructed logistic regression models to investigate which variables best predicted maternity roost use by *M. norfolkensis*. We compared roost trees to available mangroves and also to available other trees. To avoid overparamaterisation due to the small number of roosts located, our candidate models contained one or two variables per model. We avoided the use of correlated variables (Pearson’s r > 0.6 or r < -0.6) by removing the variable which was the most difficult to interpret. Our final model set contained all possible combinations of variables (Appendix C), once inter-correlated variables had been removed. We calculated the log likelihood, Akaike’s Information Criteria (Akaike 1973) value corrected for small sample size (AICc), model weight (\(W_i\)) and difference in AICc (\(\Delta\text{AICc}\)) for all models within the 95% confidence set (Burnham and Anderson 2002). The best-fitting model was considered to be the model with the lowest
AICc value. The variation explained by each model was assessed by calculating the Nagelkerke's $R^2$ value (Nagelkerke 1991) and we calculated the area under the curve (AUC) for the receiver operating characteristic (ROC) to assess the predictive capacity of the models (where value of 0.5 suggests a completely random model and 1 indicates perfect discrimination). We classified the predictive capacity of our models as poor if they had AUC values between 0.5 – 0.7; 0.7 – 0.9 as good; and > 0.9 as excellent following Psyllakis and Brigham (2006). The logistic regression analyses were conducted in R (R Development Team 2012) using the ‘AICcmodavg’ (Mazerolle 2012) and ‘verification’ (NCAR - Research Application Program 2012) packages.

4.4 Results

A total of 1057 bats, representing 10 species, were captured in the Hunter Estuary mangrove forest during the study, 21 *Chalinolobus gouldii*, six *Chalinolobus morio*, one *Miniopterus australis*, 726 *M. norfolkensis*, 17 Mormopterus species 2, 59 *Myotis macropus*, 15 *Nyctophilus geoffroyi*, 36 Scotorepens orion, 114 Scoteanax rueppellii and 62 *Vespadelus vulturnus*. Each of these species, except for *M. australis*, are hollow-roosting and could possibly roost in the mangrove forest and four of these species (*M. australis*, *M. norfolkensis*, *M. macropus* and *S. rueppellii*) are listed as vulnerable under the NSW Threatened Species Conservation Act 1995. A total of 21 *M. norfolkensis* individuals (19 lactating females and two adult males) were radio-tracked to day roosts in tree hollows, during two maternity seasons, 2009 / 2010 and 2010 / 2011. Nine *M. norfolkensis* individuals (seven lactating females and two males) were tracked during 2009 / 2010 and 12 lactating females were tracked during 2010 / 2011. Lactating female bats were tracked for 4.1 ± 0.5 days (range 0 – 7) and males for 3 and 4 days. Some bats could not be located every day, despite substantial searching in the surrounding area.
4.4.1 Roost trees and plots

In all, 40 roost trees (four male and 36 maternity) were identified during the study (Appendix E). All roost trees, except for three, were located on the Hunter River in two patches of mangrove forest (approximately 15 ha and 8 ha in size and 10 - 15m tall), within 1 km of each other (Appendix E). The average distance travelled between consecutive roosts was 143.8 ± 34.0 m (n = 21, range 16 – 557 m). All roosts were in hollows, most frequently in live grey mangrove *Avicennia marina* subsp. *australisica* (n = 34) with a smaller number located in dead stags of *A. marina* subsp. *australisica* (n = 4) and swamp oak *Casuarina glauca* (n = 2). Roost trees were located in plots with 837 ± 47.3 hollow-bearing trees / ha (range 95.5 – 1273.2 / ha), slightly less than available mangroves which had 1020.7 ± 84.6 hollow-bearing trees / ha (range 445.6 - 1559.7 / ha). Available trees in other habitats were located in plots with fewer hollow-bearing trees, 114.6 ± 43.5 trees / ha (range 31.8 - 477.5). The proportion of hollow-bearing trees in plots surrounding roosts (HBTAbundIndex of 0.78 ± 0.03) and available mangroves (HBTAbundIndex of 0.74 ± 0.05) was much higher than for plots surrounding available trees in other habitats (HBTAbundIndex of 0.07 ± 0.04). Full tree and plot measurements are provided in Appendix C.

The results of the logistic regression reflected the trends observed in the PCA (see Appendix D). The logistic regression models for maternity roosts compared to available mangroves, found that maternity roosts were more likely to occur closer to forest edges and in landscapes with urban land-use present within 1 km, than available mangroves, in the best-fitting model (RM_Mod21; Table 4-1; Appendix F). The presence of urban land-use within 1 km was the most important variable when comparing maternity roosts with available mangroves, with a relative importance index of 0.94 (Table 4-2) and this variable was included in all but one of the models within the 95 % confidence set (Table 4-1).
Table 4-1: Roost logistic regression results

Data represents results from logistic regression models of a) known roosts and available mangroves and b) known roosts and available other trees. Data presented includes Nagelkerke’s $R^2$ and the area under the curve of the receiver operator characteristic (ROC) values for models with strong support (those within 2 AICc of the best-fitting model).

<table>
<thead>
<tr>
<th>Variable(s)</th>
<th>Estimate</th>
<th>SE</th>
<th>Log(L)</th>
<th>AICc</th>
<th>$W_i$</th>
<th>$\Delta$AICc</th>
<th>ROC</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>a) Roost Vs available mangroves</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>RM_Mod21</td>
<td>-13.64</td>
<td>33.81</td>
<td>0.31</td>
<td>0</td>
<td>0.86</td>
<td>0.69</td>
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</tr>
<tr>
<td>Intercept</td>
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<td>2.89</td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distance to forest edge (log)</td>
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<td>0.86</td>
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</tr>
<tr>
<td>Presence of urban land-use within 1 km</td>
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<td>1.94</td>
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<td>0.33</td>
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<tr>
<td>Intercept</td>
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<tr>
<td>Distance to waterbody</td>
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<td>0.02</td>
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<tr>
<td>Presence of urban land-use within 1 km</td>
<td>8.80</td>
<td>3.91</td>
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<td></td>
<td></td>
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<tr>
<td>RM_Mod38</td>
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<td>34.73</td>
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<tr>
<td>Intercept</td>
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<tr>
<td>Stem density</td>
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<tr>
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<td>1.51</td>
<td></td>
<td></td>
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<td></td>
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<td><strong>b) Roosts Vs other available trees</strong></td>
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<td></td>
<td></td>
<td></td>
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<td>0.00</td>
<td>1.00</td>
<td>0.94</td>
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<tr>
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<td>0.00</td>
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<td></td>
<td></td>
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<tr>
<td>Roost tree height</td>
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<td>0.00</td>
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<td>0.99</td>
<td>0.88</td>
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<tr>
<td>Hollow-bearing tree abundance index</td>
<td>12.466</td>
<td>4.608</td>
<td>0.20</td>
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</tr>
</tbody>
</table>
Table 4-2: Relative importance indices for roost models

Relative importance indices (Burnham and Anderson 2002) calculated from AICc values of the 95% confidence set of models for Mormopterus norfolkensis maternity roosts compared to mangroves (seven models) and roost compared to other trees (six models). The highest value for each variable is highlighted in **bold**. Variables that were not included in the 95% candidate model set are indicated by a dash.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Roost Vs Mangrove</th>
<th>Roost Vs Other</th>
</tr>
</thead>
<tbody>
<tr>
<td>Distance to forest edge (log)</td>
<td>0.33</td>
<td>-</td>
</tr>
<tr>
<td>Distance to waterbody</td>
<td>0.27</td>
<td>-</td>
</tr>
<tr>
<td>Presence of freshwater wetland within 1 km</td>
<td>0.01</td>
<td>0.07</td>
</tr>
<tr>
<td>Hollow-bearing tree abundance index</td>
<td>0.02</td>
<td>1.00</td>
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<tr>
<td>Presence of mature mangrove forest within 1 km</td>
<td>0.09</td>
<td>-</td>
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<tr>
<td>Percent foliage cover</td>
<td>-</td>
<td>0.16</td>
</tr>
<tr>
<td>Roost tree height</td>
<td>0.04</td>
<td>0.44</td>
</tr>
<tr>
<td>Stem density</td>
<td>0.20</td>
<td>0.07</td>
</tr>
<tr>
<td>Presence of urban land-use within 500 m</td>
<td>-</td>
<td>0.06</td>
</tr>
<tr>
<td>Presence of urban land-use within 1 km</td>
<td>0.94</td>
<td>-</td>
</tr>
</tbody>
</table>

When compared to available trees in other habitats, maternity roosts were more likely to occur in shorter trees and to be in patches with a greater proportion of hollow-bearing trees (as described by the HBTAbundIndex in the best fitting model; RO_Mod19; Table 4-1; Appendix F). The proportion of hollow-bearing trees in the patch surrounding a maternity roost (HBTAbundIndex) was contained in every model within the 95% confidence set (Table 4-1) and was the most important variable when comparing maternity roost trees with available other trees (relative importance index of 1.00; Table 4-2). Roost tree height was not contained within any other models in the 95% confidence set and as such, is of less importance, despite being included in the best-fitting model.
Model discrimination ability was good for maternity roost trees compared to available mangroves (AUC = 0.86) and excellent for maternity roost trees compared to available other trees (AUC = 1.00). The best-fitting model for maternity roost trees compared to available trees explained a large proportion of the variation in the data (Nagelkerke’s $R^2 = 0.94$) and considerable variation (Nagelkerke’s $R^2 = 0.69$) was also explained by the best-fitting model comparing maternity roost trees with available mangroves (Table 4-1).

### 4.4.2 Roost cavities

All roosts located were in tree trunk or limb hollows at relatively low height (3.30 ± 0.38 m, range 1 - 8 m, n = 24). Entrance width varied between 1 - 18 cm at the narrowest point (3.8 ± 0.9 cm, n = 24) and between 1.5 – 50 cm at the widest point (10.1 ± 2.3, n = 24). Many roost hollows had dead heartwood fragments creating small internal spaces of less than 1.5 cm width and so were not open chambers, but did have an average depth of 53.8 ± 13.4 cm (range 10 - 150 cm). The average internal diameter at the widest point was 4.3 ± 1.0 cm and this varied between 1 - 10 cm (n = 12). Roost hollows most frequently had a southerly aspect (8); however, westerly (6), easterly (4) and northerly (2) aspects were also recorded. We also found that 33.3% of the 18 random hollow-bearing trees inspected didn’t actually have any hollows. These were trees that we visually assessed from the ground as having at least one hollow ≥ 1.5 cm, but once we inspected the hollows using a ladder and a camera, we found that they did not actually penetrate far enough into the tree to provide roosting opportunities for bats. Therefore, the available hollow-bearing tree densities, reported for the plot comparisons, are an overestimate, with one third of these trees unlikely to contain hollows.

The PCA on size and depth attributes of roost hollows indicated that there was substantial overlap with the characteristics of nearby hollows, suggesting that roost hollows were similar to the available nearby hollows (see Appendix D). Random hollows were low (2.76
During camera inspections, we observed four insectivorous bat species occupying three different hollows. We found solitary *M. macropus* and *S. orion* roosting in separate available hollows and most notably, we found both *S. rueppellii* and *M. norfolkensis* co-occupying the same hollow that was used previously by a radio-tracked lactating female *M. norfolkensis*.

### 4.4.3 Roosting groups

Observations of roosts at dusk found that bats exited 34.1 ± 1.9 min (n = 26, range 2 – 58 mins) after sunset (Appendix G), and then moved quickly out of signal range (average 6.5 ± 1.3 mins; n = 10, range 2 - 15 mins). Maternity roosting group size averaged 7.9 ± 2.3 individuals (median 4.5, range 1 – 47, n = 27) with only one male roost exit observed, being a solitary bat. It is possible that roosting groups were comprised of multiple species, as observers could not differentiate between species in flight. Lactating females had a maternity roost re-use index value of 0.69 ± 0.07, indicating a relatively low level of reuse during the short period of study. Additionally, lactating females switched roosts regularly, with roosts used for an average of 1.3 ± 0.7 consecutive nights (range 1 - 3, n = 43).

### 4.4.4 Microclimate

There were trends for relative humidity to be higher in mangroves than other habitats, and for temperature and relative humidity to be similar in roost hollows compared to available hollows and ambient mangrove locations (Figure 4-2). The nightly rate of temperature

± 0.19 m, range 0.5 - 5 m, n = 35) and 42.1 ± 6.9 cm deep, on average (range 2 - 200, n = 36). Entrance width varied between 1 - 15 cm at the narrowest point (6.8 ± 0.9 cm, n = 36) and between 1 - 20 cm at the widest point (9.0 ± 1.0, n = 36). The average internal diameter at the widest point was 5.7 ± 0.8 cm and this varied between 1.5 - 15 cm (n = 34).
change was found to be significantly different between mangroves and other habitats, with other habitats changing faster than mangroves ($t_2 = -4.60; p = 0.044$; Appendix H). There was also a trend for the daily rate of temperature change to be greater in other habitats compared with mangroves ($t_2 = 3.02; p = 0.094$; Appendix H). Nightly temperature in other habitats was more variable with a greater temperature range ($5.5 \pm 0.3 ^\circ C$) compared with mangroves ($4.0 \pm 0.1 ^\circ C$; $t_2 = 3.75; p = 0.064$). Additionally, the average 24 hour temperature of mangroves ($24.1 \pm 1.6 ^\circ C$) was significantly lower than other habitats ($24.68 \pm 1.69 ^\circ C$; $t_2 = 5.16; p = 0.036$), albeit by a small margin (mean difference $= 0.6 ^\circ C$; Appendix H).

Figure 4-2: Average hourly relative humidity and temperature graph

Average hourly relative humidity (%) and temperature ($^\circ C$) recorded over five days by USB data-loggers ($n = 2$) installed to record ambient mangrove (dashed line), mangrove *Mormopterus norfolkensis* roost hollow (black line) and nearest available mangrove hollow (grey line) and nearby Bureau of Meteorology weather stations (dotted line). Categories from each replicate were sampled concurrently, but replicates were not sampled at the same time. Mangrove sites were located within 30 m of each other and the nearest two Bureau of Meteorology stations (11 and 13 km from mangrove sample sites) were used.
4.5 Discussion

In this first study to investigate roost selection by a *M. norfolkensis* maternity colony, we found that Hunter Estuary mangrove forests are important and locally unique, having a high density of hollow-bearing trees and a stable microclimate. This is also one of few international bat-mangrove studies (see Luther and Greenberg 2009 for a review) and the first insectivorous bat study undertaken in temperate Australian mangroves. While *M. norfolkensis* is unlikely to be a mangrove-obligate (see Chapter 3), we did not expect that mangroves would be used and this study illustrates that the specialised requirements or unusual behaviour of threatened species elsewhere, may also have been overlooked in conservation strategies that have been based largely on anecdotal information. Our study suggests that the attributes of mangrove patches, such as a high proportion of hollows, were more important to lactating females than roost tree or landscape characteristics. Additionally, contrary to predictions, maternity roosting group sizes were relatively small, indicating that bats were not deriving thermoregulatory benefits from communal living. Instead, we hypothesise that individuals are gaining fitness benefits from the operation of a fission-fusion society and that mangrove forests provide benefits as roosting habitat for bats such as a high abundance of hollow-bearing trees, stable microclimate and potentially fewer predators and competitors.

4.5.1 Microclimate

The microclimate in *M. norfolkensis* maternity roost hollows was not greatly different to ambient conditions in mangrove forests, nor did *M. norfolkensis* roost in large trees or have large roosting group sizes, contrary to our predictions. This contrasts with many studies that have suggested that hollows in large trees (greater DBH and height) are more likely to be used as roosts as they are well insulated, offering thermoregulatory benefits (Lunney et al. 1988, Sedgeley and O'Donnell 1999, Law and Anderson 2000, Sedgeley 2001, Lumsden et al. 2002b, Lumsden and Bennett 2005, Ruczynski and Bogdanowicz
2008). Additionally, lactating female bats have been found to select warmer roosts as this
minimises the need for bats to enter torpor which can reduce milk production and thus
influence juvenile survival (Kerth et al. 2001, Ruczynski 2006, Law and Chidel 2007). The
temperature in roosts can be raised by up to 7 ° through communal roosting (Willis and
Brigham 2007) and this is often suggested as a reason that maternity roosts are mostly
communal and roosting groups larger in size, than at other times of year (Lumsden et al.

However, the mangrove forest ambient conditions were more stable, with temperature
being slightly less variable and changing more slowly, than other nearby habitats. There
was also a trend for humidity to be less variable and higher at night in mangrove forests
than in other habitats and this is most likely due to the presence of the nearby river. We
suggest that the mangrove forest itself provides suitable microclimate conditions to reduce
thermoregulatory costs and thus minimise the need for bats to be selective about roosts.
Particularly, the higher humidity at night (albeit only slightly higher) may reduce
dehydration in juvenile bats that are left in roosts while females forage. An analogous
situation occurs for small groups of golden-tipped bats Kerivoula papuensis roosting in
suspended nests of birds in sheltered rainforest (Law and Chidel 2004). It is also possible
that selection pressures for roosts with warm microclimates may be less in warm
temperate regions, such as the study area, than in colder climates where some of the
previous studies of roost microclimate have been undertaken (e.g. Kerth et al. 2001,
Sedgeley 2001, Ruczynski and Bogdanowicz 2008). However, pregnant and lactating
females have been found to select very hot roosts in Australia (e.g. Law and Chidel 2007),
indicating that microclimate may still exert strong selection pressures even in regions with
warm climates.
4.5.2 High patch fidelity and a fission-fusion society

Lactating female *M. norfolkensis* switched roosts often, but exhibited high patch fidelity, with all bats roosting less than 1 km from where they were trapped and relatively close to the previous roost used (143.8 ± 34.0 m). This is despite tracked bats being recorded foraging up to 10 km away each night (Chapter 6) in areas that contained other potential roosts. These characteristics of high patch fidelity and regular roost switching are common to maternity colonies and are consistent with the operation of a fission-fusion society. The use of familiar patches and those with reliable roosting habitat by bats, avoids the energetic costs associated with searching for new roosting locations and carrying offspring great distances between roosts each night. An abundance of available hollows may also facilitate the operation of a fission-fusion society by allowing regular roost switching within small patches. The mangrove forests had over 11 times more hollow-bearing trees per ha than other habitat, which is consistent with a review of roost selection studies that found regular roost switching was related to roost availability (Lewis 1995). Roost switching is thought to provide benefits to bats, including avoiding predators and reducing parasite loads (which may particularly benefit maternity colonies, as hairless juveniles are particularly susceptible). Our mangrove study site supports a substantial population of *M. norfolkensis* (as verified during trapping), despite the presence of mostly small maternity roosting group sizes. Small roosting group sizes are likely to limit the efficiency of information exchange among the colony as a whole, if most communication occurs in roosts. However, we suggest that social activity on emergence and return events during the night or on dawn may be the primary transmission time in the mangroves for information exchange among the maternity colony as a whole. Certainly, *M. norfolkensis* activity levels peaked after dusk in the mangroves and included many social calls that also occurred throughout the night (A. McConville, unpublished data 2012).
Alternative possible explanations for why mangrove forests were selected over other habitats by lactating *M. norfolkensis* include the close proximity to foraging grounds and inter-species interactions (less predation and competition pressures). Tracked bats moved quickly away from roosts that were located close to the forest edge which may have facilitated movement to foraging grounds. Indeed, radio-tracking of lactating females at night found that bats foraged primarily over a large freshwater wetland to the west of roosts (Hexham Swamp). However, bats had large home ranges and travelled up to 10 km away during the night (Chapter 6) and this suggests that close proximity to foraging habitats is unlikely to be the major driver of mangrove roost selection. Lactating females also roosted in areas with urban land-use within 1 km, which is in contrast to habitat selection studies that overlap the current study area and that found a negative association with urban land-use (Chapters 2, 5 and 6). Perhaps roosting *M. norfolkensis* are able to tolerate urban land-use in nearby landscapes, even though foraging individuals avoid it. Indeed, landscape-scale predictive mapping of *M. norfolkensis* presence-absence indicated that the mangrove forests in which bats roosted were highly suitable (probability of occurrence > 0.9), despite their close proximity to the urban centre of Newcastle (Chapter 2).

Inter-species interactions may be different in the mangroves compared to other habitats in two ways. Firstly, mangroves may have fewer predators or fewer competing hollow-dependent fauna species than other nearby habitats. Secondly, it is possible that predator and competitor abundances may be similar to other habitats, but the high density of hollow-bearing trees recorded in the mangrove forests may mean that hollows are no longer the limiting resource, reducing selection pressures. Abundant competitors have been found to be negatively associated with roost selection by insectivorous bat species (eg hollow-nesting birds; Threlfall et al. 2013a). However, whilst hollow-nesting bird species were recorded in the mangroves during this study (e.g. *Platycercus eximius*
eastern rosella and *Cormobates leucophaea* white-throated treecreeper), they appeared to be less abundant than hollow-nesting birds seen in scattered trees in farmland (Law et al. 2000). It is also possible that competition from other hollow-roosting bat species (eight additional species captured within the mangrove forest during our study) influenced *M. norfolkensis* roost selection within the mangroves. Additionally, a number of potential predators occur in habitats surrounding the mangroves such as *Felis catus* (feral cat), *Rattus rattus* (black rat), *Antechinus* sp., *Varanus varius* (lace monitor), *Trichosurus vulpecula* (brushtail possum) and arboreal snakes. However, only black rats and cats have been recorded in or near mangroves during this and related studies (A. McConville unpublished data 2012). Further study of these interspecies interactions in the mangrove forests would be insightful to fully understand the benefits that mangrove forests provide to roosting bats.

### 4.5.3 Conservation implications

The mangrove forests of temperate Australia have been overlooked as roosting habitat for insectivorous bats. The few published bat-mangrove studies are restricted to the extensive mangrove forests of Brazil (Bordignon 2006, Andrade et al. 2008) and north-western Australia (McKenzie and Rolfe 1986, McKenzie and Bullen 2012) and it is possible that the importance of mangrove forests to insectivorous bats may have also been overlooked in other regions of the world. Whist our study has focused on the importance of Hunter Estuary mangrove forests to lactating female *M. norfolkensis*, these forests are also likely to be important to other bat species. We captured a high diversity of hollow-roosting insectivorous bats within the mangrove forest, three of which were confirmed to roost in the mangroves during the study. In particular, *S. rueppellii*, another rarely captured and threatened insectivorous bat species, also has maternity roosting groups in the same two mangrove patches (B. Law unpublished data 2012). *Mormopterus norfolkensis* appears to use the mangroves primarily for roosting, moving away from
mangroves quickly and foraging elsewhere (Chapter 6). However, how other species use the mangrove forests is unknown and warrants further investigation.

Some fission-fusion bat societies have been found to be stable (Kerth et al. 2011) and maternity colonies found to be loyal to roosting areas over the long term (Willis et al. 2003, Popa-Lisseanu et al. 2008). Such long-term associations of maternity colonies would make these mangrove forests an even greater conservation priority and further work should be undertaken to determine whether *M. norfolkensis* maternity colonies, as a whole, are exclusive and stable over the medium to long term. Conservation and land management strategies need to account for the specialised requirements and potential unexpected occurrence of threatened species, rather than relying on anecdotal information. We encourage research on the importance of mangrove forests to other terrestrial fauna species, particularly in temperate regions where the stable microclimate of mangrove forests may offer particular fitness benefits.

### 4.6 Acknowledgements

We would like to thank the numerous volunteers who assisted with field work and land managers that provided property access for this project. J. Sedgeley and an anonymous reviewer provided helpful comments on this manuscript and L. Gonsalves provided feedback on an earlier version. Funding for this PhD project was awarded to AM by: The Tom Farrell Institute for the Environment; Donaldson Conservation Trust; Hunter-Central Rivers Catchment Management Authority; Royal Zoological Society of NSW (Ethel Mary Read Award); Wambo Coal Pty Ltd; and Australian Geographic Society. This project was conducted under scientific licence and animal ethics approvals, under the *Australian Code of Practice for the Care and Use of Animals for Scientific Purposes, 7th edition, 2004*. 
Appendices

A description of the study area (Appendix A), radio-tracking methods (Appendix B), candidate variables (Appendix C), principal components analyses (Appendix D), bat roost locations (Appendix E), model response plots (Appendix F), bat roost exit times (Appendix G) and temperature measurements (Appendix H) are provided below.

Appendix A: Study area

The study was conducted in the Hunter Estuary, NSW, a 3000 ha wave-dominated estuary at the Port of Newcastle and adjacent to the Newcastle central business district (32°55’36"S 151°46’44"E). The area experiences a warm temperate climate (average monthly temperatures 8.4 – 25.6 °C) and average annual rainfall of 1134 mm (Bureau of Meteorology 2012). Harp trapping of bats for radio-tracking was undertaken in the western portion of what is now a 2600 ha landmass, known as Kooragang Island. Approximately 40% is industrial land, 45% mangrove and estuarine and 15% pasture (Hamer et al. 2002). Kooragang Island is bounded by the north and south arms of the Hunter River and was formed by the artificial joining and reclamation of a number of deltaic islands that existed in the estuary. Historical reclamation works involved the disposal of river dredging spoil and industrial waste from nearby steel works (Hamer et al. 2002). The western portion of Kooragang Island was historically utilised for agriculture including cattle grazing, dairy, fruit growing, timber cutting and salt production. Most of the northern part of Kooragang Island, is part of the Hunter Wetlands National Park, a wetland of international importance under The Convention on Wetlands of International Importance 1971 (Ramsar Convention). The southern portion of Kooragang Island is heavily industrialised with coal export infrastructure being the primary land-use.

The Hunter Estuary contains high quality remnant estuarine vegetation communities, with the second largest area of mangroves (1600 ha) and the third largest area of saltmarsh
(600 ha) in NSW (Geoscience Australia 2012). The non-industrial areas of Kooragang Island are dominated by mangroves and saltmarsh, with pasture and smaller areas of freshwater wetland also occurring. The mangrove forest is largely a monoculture of *Avicennia marina* subsp. *australasica* (grey mangrove), with *Aegiceras corniculatum* (river mangrove) occasionally occurring. Mangrove forests in parts of the Hunter Estuary can form relatively tall and closed forests in comparison with the short, sparse mangrove remnants that often occur elsewhere in NSW.

Areas surrounding the Hunter Estuary have been highly modified by a long history of agriculture, coal mining and port-associated industries. The fertile floodplain of the Hunter River has been extensively cleared for agriculture with only small patches of native vegetation remaining. Hexham Swamp, a large 2500 ha wetland dominated by freshwater reeds at the time of the study, occurs to the west. Further west of Hexham Swamp are areas of undulating hills which support young dry sclerophyll forest with few hollow-bearing trees that overlay nutrient-poor soil. The most intact native vegetation occurs to the north of the estuary on the flat, sandy and nutrient poor soils of the Tomago sandbeds, which are an important local groundwater resource. However, even these areas have been subject to disturbances such as sand mining, military facilities and infrastructure.
Appendix B: Radio-tracking methods

We captured bats using harp traps (Austbat P/L, Victoria, Australia) mostly set in mangrove forests, with some placed on tracks where bush regeneration plantings of young native trees and shrubs occurred. Lactating females were identified by the presence of bare patches around enlarged nipples and / or expression of milk. We attached radio-transmitters to the dorsal surface of bats by parting the fur and gluing transmitters using adhesive glue (VetBond, 3M). Three different types of transmitters were used: LT4-337 (Titley Electronics, Ballina, Australia) with 30cm aerials during 2009; LB-2N (Holohil, Carp, Canada) with 14 cm aerials during early 2010; and a combination of LB-2N and LT6-337 (Titley Electronics) with 15 cm aerials during late 2010. Transmitters weighed between 0.35 g (Holohil) and 0.4 g (Titley Electronics), which represented ≤ 5% of the body weight of individual bats.

We conducted searches for day roosts within a 10 km radius (maximum nightly flight distance; Chapter 6) from the trapping location, with more effort expended in areas closer to the trapping location than those further away. Day roosts were located from a vehicle, boat or on foot using hand-held directional antennae. Signals from roosts were detectable from 20 - 1000 m depending on roost height, topography and vegetation density. The position of the bat within the tree was estimated using triangulation and signal strength and was later confirmed by observations at dusk. Roost trees were located daily for the battery life of the transmitter or until the transmitter fell off.

Roost trees were watched by observers from 30 min before dusk to 1 h after dusk. Mangrove roosts were difficult to observe due to the relatively dense canopy cover and some trees were not accessible at high tide and were not watched. Where bats were observed to exit, the number of bats that emerged and entered were recorded. We subtracted the number of bats that entered the roost during the watching period from the
total colony size. Each observer had a radio-telemetry receiver to identify tracked bats and a bat detector (Anabat SD1, Titley Electronics, Balina, Australia) placed on the ground with microphones angled at 45 degrees to record bat echolocation calls. Colony size, time of tracked bat exit, exit direction, hollow aspect and various hollow dimensions were recorded.
### Appendix C: Candidate variables

#### Table S1: Candidate tree, plot and landscape variables from roost trees, available mangroves and available other trees

Available other trees were selected from sites from each of the major vegetation types present within 10 km radius of the roosts. * maternity roosts only, males were excluded from this analysis. Plot refers to the 10 m radius plot surrounding a focal centre tree which was either a roost or available hollow-bearing tree. Values are mean ± SE (range) or frequency of ordinal values.

<table>
<thead>
<tr>
<th>Candidate Variables</th>
<th>Scale</th>
<th>Description</th>
<th>Roosts (n = 34)*</th>
<th>Available Mangroves (n = 15)</th>
<th>Other trees (n = 10)</th>
</tr>
</thead>
<tbody>
<tr>
<td>DBH</td>
<td>Tree</td>
<td>Diameter at breast height over bark (DBH) of tree (cm)</td>
<td>24.0 ± 1.2 (14.3 – 41.4)</td>
<td>24.4 ± 1.6 (14.3 – 33.7)</td>
<td>41.3 ± 6.1 (15.9 – 90.1)</td>
</tr>
<tr>
<td>TopoPos</td>
<td>Tree</td>
<td>Topographic position: 1- 8 scale 1. hilltop 2. ridge 3. upper slope 4. mid slope 5. lower slope 6. flat 7. gully 8. creek 9. depression</td>
<td>7.9 ± 0.1 (6 – 8)</td>
<td>8 ± 0 (8) (6 – 8)</td>
<td>5.4 ± 0.36 (4 – 6)</td>
</tr>
<tr>
<td>DecayClass</td>
<td>Tree</td>
<td>Tree decay class (following Gibbons et al. 2000), 1-8 scale with 1 being healthy with no hollows and 8 being highly decayed</td>
<td>2.6 ± 0.3 (2 – 8)</td>
<td>2 ± 0 (2)</td>
<td>2.9 ± 0.5 (2 – 7)</td>
</tr>
<tr>
<td>PFC</td>
<td>Plot</td>
<td>Percent foliage cover (PFC), converted to ordinal categories for analyses (1 is &lt;= 30; 2 is &gt; 30 and &lt;= 50; 3 is &gt; 50 %)</td>
<td>50.9 ± 1.7 (10 – 65)</td>
<td>54 ± 1.4 (50 – 65)</td>
<td>36.5 ± 2.8 (20 – 50)</td>
</tr>
<tr>
<td>RoostTreeHt</td>
<td>Tree</td>
<td>Roost Tree Height (m)</td>
<td>9.8 ± 0.6 (3 – 15)</td>
<td>12.5 ± 0.4 (10 – 15)</td>
<td>14.1 ± 1.1 (9 – 20)</td>
</tr>
<tr>
<td>DistancetoWB_cat</td>
<td>Tree</td>
<td>Distance to nearest waterbody with open water for drinking. Ordinal category: 1 is ≤ 100 m; 2 is 100 – 1000 m; 3 is ≥ 1000 m.</td>
<td>1 = 28 = 1 = 9</td>
<td>2 = 6 (2 = 6)</td>
<td>2 = 3 (2 = 3)</td>
</tr>
<tr>
<td>DistanceForEdge</td>
<td>Tree</td>
<td>Distance to forest edge (m)</td>
<td>47.0 ± 3.6 (0 – 95)</td>
<td>102.8 ± 16.4 (35 – 244)</td>
<td>114.4 ± 37.5 (10 – 330)</td>
</tr>
<tr>
<td>Stem Density_ha</td>
<td>Plot</td>
<td>Stem density per hectare derived from the number of stems &gt; 3 cm DBH in each plot</td>
<td>1062.6 ± 78.3 (159.2 – 2387.3)</td>
<td>1468.5 ± 175.3 (668.5 – 2864.8)</td>
<td>1833.5 ± 291.8 (382.0 – 2896.7)</td>
</tr>
<tr>
<td>HBTDensity_ha</td>
<td>Plot</td>
<td>Hollow-bearing tree density per hectare derived from number of hollow-bearing stems per plot</td>
<td>837.0 ± 47.3 (95.5 – 1273.2)</td>
<td>1020.7 ± 84.6 (445.6 – 1559.7)</td>
<td>114.6 ± 43.5 (31.8 – 477.5)</td>
</tr>
<tr>
<td>Candidate Variables</td>
<td>Scale</td>
<td>Description</td>
<td>Roosts (n = 34)*</td>
<td>Available Mangroves (n = 15)</td>
<td>Other trees (n = 10)</td>
</tr>
<tr>
<td>---------------------</td>
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<td>------------------------------------------------------------------------------</td>
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<td>---------------------</td>
</tr>
<tr>
<td>HBTAbundindex</td>
<td>Plot</td>
<td>Number of hollow-bearing trees / number of stems in plot. Converted to ordinal category for PCA (1 is &lt;= 0.5; 2 is &gt; 0.5 and &lt; 0.85; 3 is &gt;= 0.85)</td>
<td>0.78 ± 0.03 (0.18 – 1)</td>
<td>0.74 ± 0.05 (0.22 – 0.97)</td>
<td>0.07 ± 0.04 (0 – 0.42)</td>
</tr>
<tr>
<td>AvgOfDecayClass</td>
<td>Plot</td>
<td>Average of stem decay class within plot</td>
<td>2.3 ± 0.1 (1.4 – 3.3)</td>
<td>2.4 ± 0.1 (1.8 – 3.1)</td>
<td>1.3 ± 0.1 (1.1 – 1.7)</td>
</tr>
<tr>
<td>AvgOfDBH</td>
<td>Plot</td>
<td>Average of stem DBH (cm) within plot</td>
<td>19.8 ± 0.9 (10.1 – 34.2)</td>
<td>17.8 ± 1.5 (8.8 – 27.4)</td>
<td>14.1 ± 3.1 (7.9 – 40.3)</td>
</tr>
<tr>
<td>FW_500_pres</td>
<td>Landscape</td>
<td>Binary freshwater wetland category (&gt; 5 % freshwater wetland = 1; &lt; 5 % freshwater wetland = 0) within two buffers of tree (500 m; 1 km radii). Wetland boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
<tr>
<td>FW_1000_pres</td>
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<td>Binary freshwater wetland category (&gt; 5 % freshwater wetland = 1; &lt; 5 % freshwater wetland = 0) within two buffers of tree (500 m; 1 km radii). Wetland boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
<tr>
<td>VEG_500_pres</td>
<td>Landscape</td>
<td>Binary vegetation category (&gt; 5 % vegetation = 1; &lt; 5 % vegetation = 0) within two buffers of tree (500 m; 1 km radii). Woody vegetation only. Boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
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<tr>
<td>VEG_1000_pres</td>
<td>Landscape</td>
<td>Binary vegetation category (&gt; 5 % vegetation = 1; &lt; 5 % vegetation = 0) within two buffers of tree (500 m; 1 km radii). Woody vegetation only. Boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
<tr>
<td>MATMANG_500_pres</td>
<td>Landscape</td>
<td>Binary mature mangroves category (&gt; 5 % mature mangroves = 1; &lt; 5 % mature mangroves = 0) within two buffers of tree (500 m; 1 km radii). Mature mangroves only. Boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
<tr>
<td>MATMANG_1000_pres</td>
<td>Landscape</td>
<td>Binary mature mangroves category (&gt; 5 % mature mangroves = 1; &lt; 5 % mature mangroves = 0) within two buffers of tree (500 m; 1 km radii). Mature mangroves only. Boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
<tr>
<td>URB_500_pres</td>
<td>Landscape</td>
<td>Binary urban land-use category (&gt; 5 % urban land-use = 1; &lt; 5 % urban land-use = 0) within two buffers of tree (500 m; 1 km radii). Boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
<tr>
<td>URB_1000_pres</td>
<td>Landscape</td>
<td>Binary urban land-use category (&gt; 5 % urban land-use = 1; &lt; 5 % urban land-use = 0) within two buffers of tree (500 m; 1 km radii). Boundaries digitised in GIS from aerial photography</td>
<td>500 1 = 0 0 = 34 1000 1 = 21 0 = 13</td>
<td>500 1 = 0 0 = 15 1000 1 = 1 0 = 14</td>
<td>500 1 = 0 0 = 14 1000 1 = 0 0 = 9</td>
</tr>
</tbody>
</table>
Appendix D: Principle Components Analyses

Methods
We conducted a Principle Components Analysis (PCA), using JMP (SAS Institute, version 9.0) to assess the similarity among maternity roosts selected by *M. norfolkensis* compared to available mangroves and available other trees using tree, plot and landscape characteristics (Table S1). Additionally, we conducted a separate PCA to assess the similarity of roost hollows with adjacent available hollows using entrance and internal dimensions. A correlation matrix on normalised data was used in both of the PCA.

Results
The PCA on attributes of roosts selected by lactating females indicated that there was overlap with available mangroves and a high level of separation from available other trees (Figure S1). The first three axes accounted for 63.9 % of the variation in the data, with Component 1 explaining the most variation (36.6 %). Roosts and available mangroves grouped together higher on the Component 1 axis than other trees (Figure S1), with the most important factors describing roosts and available mangroves being the presence of mature mangroves within 500 m, close proximity to water bodies and in patches with greater decay and a high proportion and density of hollow-bearing trees. Additionally, of less importance, shorter trees in plots with greater canopy cover and with freshwater wetland, mature mangroves and urban land-use within 1 km describes roosts and available mangroves on the Component 1 axis.

Roosts were also grouped lower on the Component 3 axis away from available mangroves (Figure S1), which indicates roost trees were closer to the forest edge than available mangroves. Roost trees were also shorter and in plots that had a lower proportion of hollow-bearing trees, less foliage cover, but had freshwater wetland and urban land-use within 1 km than available mangroves on the Component 3 axis.
The PCA on size and depth attributes of maternity mangrove hollows indicated that there was substantial overlap with nearby hollows (Figure S2), suggesting that maternity hollows were similar to adjacent available hollows. The first two axes accounted for 85.3% of the variation in the data, with Component 1 explaining the most variation (53.7%).
Figure S1: Plots of the first three principal components using tree, patch and landscape variables of known roosts (X), random mangrove (square) and other tree (circle)

The first three axes account for 63.9% of the variation on the data. See Table S1 for explanation of candidate variables.
Figure S2: Plot of the first two principal components using hollow depth, hollow internal area and hollow entrance area of known *Mormopterus norfolkensis* maternity roosts (X) and available mangrove hollows (square)

The first two axes account for 85.3% of the variation on the data.
Appendix E: Roost location

Figure S4: Location of maternity roosts (green triangle) and male roosts (orange circle) on the south arm of the Hunter River, NSW
Figure S5: Mature mangrove forest in the Hunter Estuary, NSW
Figure S6: *Mormopterus norfolkensis* roost hollow in mature mangrove forest, Hunter Estuary, NSW.
Appendix F: Model response plots

a)
Figure S7: Roost logistic regression model response plots

Partial-plots of the relationship between probability of *Mormopterus norfolkensis* maternity roost occurrence and environmental variables for the best fitting models comparing a) roosts to mature mangroves; and b) roosts to other trees. The dashed lines indicate 95% confidence intervals. The x-axis represents the range of
values sampled for each environmental variable. Over-plotting of multiple points is visualized by increasingly darker shades of grey.
Appendix G: Bat roost exit times

Student’s t-tests were used to compare the emergence time of tracked *M. norfolkensis* and the time that the first bat exited the roost to the time of first bat activity in the mangroves obtained from bat echolocation calls recorded using ultrasonic bat detectors (Anabat SD1, Titley Electronics, Balina, Australia).

Tracked bats emerged 34.1 ± 1.9 minutes (n = 26, range 2 – 58 mins) after sunset, which was significantly later than the first bat activity recorded on bat detectors at 13.5 ± 1.5 minutes after sunset (t_{83} = 9.08, p < 0.001; n = 35, range 0 – 36 mins; Figure S8). Additionally, the first bats out of the roosts exited significantly later (30.5 ± 1.5 minutes after sunset, n = 25) than the first bat activity recorded (t_{83} = 7.42, p < 0.001). Bats were then usually observed to move quickly out of signal range (average 6.5 ± 1.3 mins; n = 10, range 2 - 15 mins). We occasionally observed solitary bats entering roosts on dusk, which was usually followed by audible noises from bats already located within the roost and then a single bat exiting shortly afterwards.
Figure S8: Emergence times for tracked *Mormopterus norfolkensis* (X), first bat activity (square) and start of *M. norfolkensis* roost emergence (diamond)

Emergence data from the Hunter Estuary mangroves during 2009 - 2011 are combined. Dark and light lines represent sunset and civil twilight times respectively.
Appendix H: Temperature measurements

Table S2: Average temperature measurements

Average temperature (± SE) recorded in mangroves (n = 3) compared with other habitats (n = 3). Temperature was recorded over a 5-day period, with results presented as mean temperature in three time blocks: a) 24hr (1:00 – 24:00 h), b) day (07:00 – 19:00 h) and c) night (20:00 – 06:00 h). Paired t-tests were used to summarise differences (df = 2) and significant differences (*) reported at α = 0.05.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Mangroves</th>
<th>Other Habitat</th>
<th>t</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>a) 24 hour</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>24.05 ± 1.63</td>
<td>24.68 ± 1.69</td>
<td>5.16</td>
<td>0.0355*</td>
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<tr>
<td>Minimum</td>
<td>20.31 ± 1.42</td>
<td>18.81 ± 0.67</td>
<td>-1.81</td>
<td>0.2126</td>
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<tr>
<td>Maximum</td>
<td>27.60 ± 1.94</td>
<td>31.58 ± 2.96</td>
<td>2.71</td>
<td>0.1131</td>
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<tr>
<td>Range</td>
<td>7.29 ± 1.15</td>
<td>12.77 ± 2.30</td>
<td>2.39</td>
<td>0.1393</td>
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<tr>
<td>Rate of Change</td>
<td>-0.06 ± 0.02</td>
<td>-0.06 ± 0.01</td>
<td>-0.11</td>
<td>0.9218</td>
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<tr>
<td><strong>b) Day</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>25.06 ± 1.53</td>
<td>27.26 ± 2.08</td>
<td>2.86</td>
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<td>Minimum</td>
<td>21.24 ± 1.47</td>
<td>19.79 ± 0.47</td>
<td>-1.31</td>
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<tr>
<td>Maximum</td>
<td>27.53 ± 1.94</td>
<td>31.58 ± 2.96</td>
<td>2.65</td>
<td>0.1177</td>
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<tr>
<td>Range</td>
<td>6.28 ± 1.47</td>
<td>11.79 ± 2.5</td>
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<td>Rate of Change</td>
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<td>0.40 ± 0.02</td>
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<tr>
<td><strong>c) Night</strong></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>22.77 ± 1.65</td>
<td>21.50 ± 1.04</td>
<td>-1.90</td>
<td>0.1971</td>
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<tr>
<td>Minimum</td>
<td>21.22 ± 1.72</td>
<td>19.56 ± 1.00</td>
<td>-2.16</td>
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<td>Maximum</td>
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<td>25.05 ± 1.23</td>
<td>-0.43</td>
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<td>Range</td>
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<td>5.49 ± 0.28</td>
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<tr>
<td>Rate of Change</td>
<td>-0.42 ± 0.03</td>
<td>-0.63 ± 0.01</td>
<td>-4.60</td>
<td>0.0441*</td>
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</tbody>
</table>
Appendix I: Radio-tracking details for individual bats

Table S3: Details for radio-tracked *Mormopterus norfolkensis*

Details for radio-tracked *Mormopterus norfolkensis* including date of capture, number of roost trees used, number of days located and total number of days tracked. MA = male adult; FL = lactating female

<table>
<thead>
<tr>
<th>Bat ID</th>
<th>Sex</th>
<th>Date captured</th>
<th>Number days tracked</th>
<th>Number of days located</th>
<th>Number roosts used</th>
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<tr>
<td>M5</td>
<td>MA</td>
<td>10/12/2009</td>
<td>3</td>
<td>3</td>
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<tr>
<td>M6</td>
<td>FL</td>
<td>16/12/2009</td>
<td>3</td>
<td>1</td>
<td>1</td>
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<td>M7</td>
<td>MA</td>
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<td>2</td>
<td>2</td>
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<td>FL</td>
<td>16/12/2009</td>
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<td>3</td>
<td>2</td>
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Chapter 5

Are regional habitat models useful at a local-scale? A case study of threatened and common insectivorous bats in south-eastern Australia

Anna McConville, Bradley Law and Michael Mahony

PLOS ONE
Chapter 5 Are regional habitat models useful at a local-scale? A case study of threatened and common insectivorous bats in south-eastern Australia

5.1 Abstract

Habitat modelling and predictive mapping are important tools for conservation planning, particularly for lesser known species such as many insectivorous bats. However, the scale at which modelling is undertaken can affect the predictive accuracy and restrict the use of the model at different scales. We assessed the validity of existing regional-scale habitat models at a local-scale and contrasted the habitat use of two morphologically similar species with differing conservation status (*Mormopterus norfolkensis* and *Mormopterus* species 2). We used negative binomial generalised linear models created from indices of activity and environmental variables collected from systematic acoustic surveys. We found that habitat type (based on vegetation community) best explained activity of both species, which were more active in floodplain areas, with most foraging activity recorded in the freshwater wetland habitat type. The threatened *M. norfolkensis* avoided urban areas, which contrasts with *M. species 2* which occurred frequently in urban bushland. We found that the broad habitat types predicted from local-scale models were generally consistent with those from regional-scale models. However, threshold-dependent accuracy measures indicated a poor fit and we advise caution be applied when using the regional models at a fine scale, particularly when the consequences of false negatives or positives are severe. Additionally, our study illustrates that habitat type classifications can be important predictors and we suggest they are more practical for conservation than
complex combinations of raw variables, as they are easily communicated to land managers.

5.2 Introduction

Habitat models (Guisan and Zimmermann 2000, Elith et al. 2006) and predictive mapping (Jaberg and Guisan 2001, Wintle et al. 2005) have been successful at classifying habitat of wide-ranging or lesser known species. Providing such models are validated (Elith et al. 2002), predictive maps can be used to direct conservation effort, identify areas for survey and research, and investigate disturbance effects (Ferrier et al. 2002, Gibson et al. 2004, Greaves et al. 2006, Rhodes et al. 2006, Manning et al. 2007). Habitat selection operates at many spatial scales (Krebs 2001) and habitat modelling studies should select a scale that is appropriate to both the species and the model’s intended use (Franklin 2009). Habitat models prepared across the entire distribution of a species are likely to describe only broad patterns, whilst models based on a single population or family group are likely to contain location-specific variables that are unable to be extrapolated to broader scales (Razgour et al. 2011b). Both broad- and fine-scale habitat models have value. For example, landscape-scale studies are useful to identify trends in habitat use by functional groups, such as the use of urban areas by insectivorous bats (Avila-Flores and Fenton 2005, Hourigan et al. 2006, Threlfall et al. 2011). Additionally, studies that are undertaken at a fine-scale may identify factors that are important for conservation management that were unable to be identified from broader studies (Watrous et al. 2006, Brambilla et al. 2009). For land managers that may use habitat model predictions for land-use planning, it is essential to know whether models may be accurately applied to other spatial scales.

The factors that influence habitat use by a species are poorly understood for many species of conservation concern. Yet this information is fundamental to understanding
threatening processes and to the development of adequate conservation strategies. *Mormopterus norfolkensis* Gray, 1839 is a hollow-roosting insectivorous bat species of which little is known. It occurs on the east coast of Australia and is listed as vulnerable under the New South Wales (NSW) *Threatened Species Conservation Act 1995* and as vulnerable C1 under the International Union for Conservation of Nature red list (IUCN 2009). A morphologically similar species, *Mormopterus* species 2 (Adams et al. 1988), occurs in sympatry with *M. norfolkensis* in some parts of its range. Both species have morphological characteristics (high aspect ratio and high wing loading) and echolocation call designs (low frequency) indicative of species’ that are adapted to foraging in open habitats (McKenzie and Rolfe 1986, Norberg and Rayner 1987, Rhodes 2002, Adams et al. 2009). Regional-scale habitat models based on systematically collected presence / absence data have been previously developed for both species (hereafter referred to as the ‘regional-scale models’) and indicate differences between the species in relation to the use of urban and floodplain habitats (Chapter 2). However, in a similar manner to the effect of false negatives on habitat models (Tyre et al. 2003), it is possible that models created using presence / absence data inflate the importance of (and possibly over-predict) habitat for highly mobile species by assigning the same value to a site which had continuous foraging activity as a site where only a single animal passed by (a false positive).

The aim of this study was to determine if local-scale habitat associations, based on an index of habitat use (activity levels), are consistent with the predictions of regional-scale habitat models that were developed using presence / absence data. Additionally, we aimed to explore whether morphologically similar and sympatric species use different habitats by contrasting habitat models of *M. norfolkensis* with those of *M. species 2*. This was achieved with systematic local acoustic surveys in a variety of habitats, centred on a large roosting population of *M. norfolkensis* and nested within the extent of the previous
regional-scale habitat modelling. We calculated different accuracy measures to explore
model fit and predicted that if habitat use at a local-scale is consistent with regional-scale
models (Chapter 2) both species would be more likely to use floodplain habitats and *M.
norfolkensis* would be less likely to occur in urban areas.

5.3 Methods

5.3.1 Study area

The Hunter Estuary enters the Pacific Ocean at the Port of Newcastle (32°55’36”S
151°46’44”E) on the east coast of Australia (Figure 5-1). The area experiences a warm
temperate climate (average monthly temperatures 8.4 – 25.6 °C) and average annual
rainfall of 1134 mm (Bureau of Meteorology 2012). The Hunter Estuary contains high
quality estuarine vegetation communities, with the second largest area of mangroves
(1600 ha) and the third largest area of saltmarsh (600 ha) in NSW (Geoscience Australia
2012). Areas surrounding the Hunter Estuary have been highly modified by a long history
of agriculture, coal mining and port-associated industries. The fertile floodplain of the
Hunter River has been extensively cleared for agriculture with only small patches of native
vegetation remaining and the construction of flood levies has altered the pattern of river
flooding. Hexham Swamp, a large 2500 ha wetland dominated by freshwater vegetation
(since the study, floodgates have been opened to re-establish tidal flushing and more
estuarine conditions) and generally lacking trees, occurs in the west of the study area.
Further west of Hexham Swamp the topography comprises low undulating hills with
moderately nutrient-poor soils (Mattei 1995) that support remnant dry sclerophyll forest
vegetation (Figure 5-1), which is typically young with few hollow-bearing trees
(A. McConville pers. obs.). The most intact native vegetation (low open forest, woodland
and heath) occurs to the north of the estuary on the flat, sandy and nutrient-poor soils of
the Tomago sandbeds, which overlay a groundwater resource that is used locally. This
area has been subject to disturbances such as sand mining and construction and the operation of military facilities and other infrastructure.

Figure 5-1: Study area location

Sample sites (orange circle), woody vegetation cover (green shading), mature mangrove forest (black shading), freshwater wetland (blue cross hatch), urban land-use (brown stipple) and major water bodies (blue shading) within 15 km.

5.3.2 Study design and site selection

We used a 15 km buffer around two patches of mangroves in the Hunter Estuary that contained multiple maternity roosts and where over 700 *M. norfolkensis* have been trapped (Chapter 4), to delineate the study area (approximately 76,500 ha; Figure 5-1), as the maximum recorded movement of *M. norfolkensis* over one night is 10 km (Chapter 6). We centred the study area on a large roosting population of *M. norfolkensis* to focus the study at a local-scale where we knew the species was active. This also assisted with
obtaining sufficiently high activity levels to use as a response variable in statistical analyses. We classified the study area into ‘habitat type’ categories based on vegetation communities and human land-use. Habitat type categories were: freshwater wetland, dry sclerophyll forest, dry sclerophyll forest on sand; urban bushland; urban matrix; mature mangrove forest; and swamp oak forest. Small, distinct habitat types (e.g. coastal heath) and those that merged with other communities (e.g. saltmarsh) were not sampled. The freshwater wetland category included reed and rush vegetation that lacked tree cover and were dominated by *Typha bonariensis* or *Phragmites australis* with some open water. The dry sclerophyll forest habitat type consisted of open forest and woodland vegetation that was usually over 20 m in height, occurred on undulating slopes and was dominated by *Corymbia maculata* (spotted gum) and *Eucalyptus fibrosa* (broad-leaved ironbark) with *E. punctata* (grey gum) occurring occasionally. The dry sclerophyll forest on sand habitat type was located on mostly flat, sandy plains and consisted of low (10 - 20 m height) open forest and woodland vegetation communities dominated by on flat terrain by *E. haemastoma* (scribbly gum) and *Angophora costata* (smooth-barked apple) or *E. parramattensis* subsp. *decadens* (Earp’s gum) and by *E. pilularis* (blackbutt) on sandy rises. Urban bushland consisted of remnants (8 - 80 ha) of dry sclerophyll forest that were completely surrounded by the matrix of suburban and industrial development. Urban matrix habitat was located in backyards in well-established suburbs of Newcastle and contained scattered ornamental trees and shrubs. Mangrove forest was mostly a monoculture of *Avicennia marina* subsp. *australisca* (grey mangrove) and the swamp oak forest habitat type consisted of small remnant patches (0.4 - 5 ha) dominated by *Casuarina glauca* (swamp oak). A total of 47 sites were sampled, consisting of nine freshwater wetland, eight dry sclerophyll forest, eight dry sclerophyll forest on sand; five swamp oak forest, five urban bushland, five urban matrix, and seven mangrove forest habitat type sites. Sites were not selected randomly, but rather to achieve a spread of
sites within the available habitat type and across the study area. Sites were separated by at least 250 m and were located on both private and public land.

5.3.3 Bat call sampling

Sampling was undertaken between 25 January and 25 February 2011, following the maternity season (after weaning, when young are flying). One bat detector (Anabat II with CFZCAIM or Anabat SD1, Titley Electronics, Balina, Australia) was placed at each sample site to passively record for two entire nights. Up to seven sites were sampled on any one night and 10 different detectors were used during the study. Microphones were housed in a 1 m tall plastic pipe to provide some weather protection and were aimed up at a 45° angle. Detectors were set in the open, or aimed along flyways and away from vegetation in forested areas to minimise sound attenuation. Surveys were not conducted on windy or rainy nights or on a full moon to avoid any possible effect of these factors on bat activity (de Oliveira 1998). Bat detectors were calibrated prior to sampling to ensure that detector sensitivity was similar across all detectors. The average minimum nightly temperature (sourced from the nearest weather station) was calculated for each site and included as candidate variables in analyses. Rainfall and wind speed data were also collected from nearby weather stations and were used to verify that substantial rainfall or high winds were not experienced during sampling.

We developed a filter in AnalookW (Corben 2008) to reduce the number of call sequences for analysis by excluding most non-<i>Mormopterus</i> calls. The following filter parameters were used: smoothness = 30; body over 1000ms; F<sub>c</sub> = 22-39 kHz; S<sub>c</sub> = -15-20 OPS; and D<sub>c</sub> = 3.5 – 20 ms. We tested the performance of the filter on a reference library of <i>M. norfolkensis</i>, <i>M. species 2</i> and <i>M. species 4</i> calls (n = 104) and found that 95.4% were correctly identified. The reference calls that were excluded using the filter were reviewed and assessed to be poor quality recordings, mostly from old tape units. To test the filter
performance on remotely recorded data, we then ran the filter through manually identified calls from one of the survey sites. In this test, the filter correctly identified 99.4% of calls (n = 639) as *Mormopterus* spp, and we considered this to be an acceptably low level of misclassification.

Bat calls that passed the *Mormopterus* filter were then manually identified using AnalookW to view calls and extract parameters. A key to the identification of *Mormopterus* spp. in the study area was developed based on a broader NSW guide (Pennay et al. 2004) to make the process less subjective. Each pass (call sequence) was assigned to one of three categories - definite, probable or unidentified, according to the confidence with which an identification could be made (Mills et al. 1996). During manual identification, feeding buzzes were identified by their characteristic shape (Pennay et al. 2004). Passes that did not contain any search phase pulses or contained fewer than three pulses were not considered. Definite and probable identifications were included in the analyses. We used total activity (number of passes) recorded for each species, at each site (n = 2 nights), to create habitat models. Additionally, we calculated average nightly bat activity (number of passes per night) to allow comparison with other studies and calculated foraging activity (percentage of passes that contained a feeding buzz) for each *Mormopterus* spp. at each site.

### 5.3.4 Landscape variables

We used ArcGIS (version 9.3, ESRI, Redlands, CA, USA) to derive a number of environmental variables describing the sample sites and surrounding area. The slope and elevation layers were derived from a 25 m digital elevation model (DEM), which was interpolated (drainage enforced) using 10 m contours and drainage lines and regions from 1:25,000 topographic maps. We created a vegetation coverage layer from regional vegetation mapping (House 2003) with non-woody vegetation types removed
(i.e. grassland and reeds). We also created a freshwater wetland layer consisting of communities dominated by reeds (both in a natural state and used for cattle grazing) as previous predictive mapping indicated that freshwater wetlands were suitable (typically > 0.5 probability of occurrence; Chapter 2). We excluded forested wetland types as they occupied only a relatively small portion of the study area. The urban land-use layer was based on NSW Government land-use classification mapping (NSW Landuse) obtained from the NSW Office of Environment and Heritage and the soil type was based broadly on soil landscape mapping (Mattei 1995). The line-work of all layers was revised following a review of recent aerial photography (ESRI, Redlands, CA, USA, 2001) and to ensure that all layers were consistent. We used ArcGIS with the Patch Analyst extension (version 4.0 Rempel et al. 2011) to create a series of patch size and configuration variables from the vegetation dataset (Appendix A) broadly following McGarigal and McComb (McGarigal and McComb 1995). A patch was considered to be > 10 trees together with a minimum area of 500 m². Core areas were considered to be > 100 m from the vegetation edge.

To determine the spatial scale at which habitat use was occurring, we created a series of concentric buffers at 250 m, 500 m, 1 km, 2.5 km and 5 km around each sample site. We then quantified the amount and configuration of environmental variables (Appendix A) within each buffer for each site. Many of the environmental variables calculated from buffers had highly skewed distributions that could not be transformed to a normal distribution, so we converted these to ecologically relevant categories for use in the analyses (Appendix A).

5.3.5 **Statistical analyses**

We used R (R Development Core Team 2012) to prepare models and undertake other statistical analyses using the MASS (Venables and Ripley 2002), PresenceAbsence (Freeman and Moisen 2008), verification (NCAR - Research Application Program 2012)
and visreg (Breheny and Burchett 2012) packages. We use generalised linear models (GLMs) with a negative binomial distribution (glm.nb from MASS package) to model the activity of *M. norfolkensis* and *M. species 2* against environmental variables (Appendix A). We excluded the mangrove forest sites from statistical analyses as the very high activity levels were overly influential, most likely due to the use of mangroves as roosts (Chapter 4). We first constructed univariate models for each explanatory variable and species. We then ranked the univariate models by Akaike Information Criterion value corrected for small sample size (AICc; Akaike 1973), with the highest ranking model having the lowest AICc. To reduce the number of variables offered to final candidate models, the variables within the four highest ranking univariate models were then selected for each species for consideration in the final candidate models. We avoided the use of correlated variables (Pearson’s r > 0.6 or r < -0.6) in final candidate models by removing the variable which was the most difficult to interpret. Where landscape buffer variables of the same category were high ranking only the highest-ranking buffer value was selected for consideration in the final models (Burnham and Anderson 2002). The final model set contained all combinations of the four highest-ranking and non-correlated variables for each species. The habitat type variable was not combined with other variables to minimise the potential for model overparameterisation.

We then calculated AICc and model weight (which is interpreted as the likelihood that the model is the best among the 95 % confidence model set; Burnham and Anderson 2002) for each of the final candidate models. The best-fitting model for each species was considered to have the lowest AICc ranking and any model within two AICc points of the top model was considered to have strong support (Burnham and Anderson 2002). The variation explained by each model was assessed by calculating the Nagelkerke’s $R^2$ value (Nagelkerke 1991). Additionally, to assess the predictive capacity of the models we calculated the area under the curve (AUC) for the receiver operating characteristic (ROC)
(Fielding and Bell 1997, Pearce and Ferrier 2000) from the presence / absence records of each species. An AUC value of 0.5 suggests a completely random model and 1 indicates perfect discrimination. The traditional academic point system (Swets 1988) was used as a rough guide for classifying the discrimination ability of each model with AUC values (Thuiller et al. 2003) where AUC values under 0.7 are considered poor, values of 0.7 to 0.8 are rated as fair, 0.8 to 0.9 as good and those over 0.9 as excellent. Model residuals were checked for spatial autocorrelation using Moran’s I (Moran 1950). We plotted partial response curves for the best-fitting models using the conditional response option in the visreg package (Breheny and Burchett 2012). We used a chi-square goodness of fit test to compare percentage foraging activity between habitat types and we excluded habitat types with < 2 feeding buzzes from this analysis (urban matrix and urban bushland for *M. norfolkensis* and urban matrix, urban bushland and swamp oak forest for *M. species 2*). All results are reported as mean ± standard error.

### 5.3.6 Validation of regional-scale habitat models

We also used data collected during this study to assess the predictive ability of previous regional-scale habitat mapping undertaken for *M. norfolkensis* and *M. species 2* within the study area (Chapter 2). The regional-scale habitat models were prepared from data collected using bat detectors set in a similar manner as this study (mostly for two nights and aimed along flyways and tracks), within an overlapping study area of 770,000 ha (Chapter 2). The regional predictive maps were created from the best-fitting generalised linear mixed models of presence / absence data (Chapter 2). See Chapter 2 for full methodology used to create regional habitat models.

We used the AUC for the ROC of regional-scale predictions against the local-scale dataset as a threshold-independent measure of predictive accuracy. To do this, we extracted the probability of occurrence value from the regional-scale predictive mapping
(Chapter 2) for each of our sample sites in ArcGIS and converted the activity levels to presence / absence records. Additionally, we used threshold-dependent confusion matrices to calculate the following accuracy measures to describe model predictive ability: sensitivity, specificity, prevalence, correct classification rate (CCR), kappa and true skill statistic (see Fielding and Bell 1997, Allouche et al. 2006). We calculated the occurrence threshold for the confusion matrices as the probability of occurrence value from the regional-scale mapping where sensitivity was equal to specificity for the local-scale data. The kappa and TSS accuracy measures range from −1 to +1, where a value of 1 indicates perfect agreement and values of zero or less indicate a performance no better than random (Cohen 1960, Allouche et al. 2006). Kappa values of 0 – 0.4 are considered to indicate slight to fair model performance, values of 0.4 – 0.6 moderate, 0.6 – 0.8 substantial and 0.8 – 1.0 almost perfect (following Landis and Koch 1977) and this ranking was also applied to TSS values.

### 5.3.7 Ethics statement

This project was conducted under scientific licence (S132C under National Parks Act 1974; S12460) and as we used only indirect survey methods (ultrasonic detectors to record bat calls) we did not require animal ethics approval.

### 5.4 Results

In all, 5981 *M. norfolkensis* and 854 *M. species 2* passes were recorded, including 402 *M. norfolkensis* (6.7 % of total calls) and 58 *M. species 2* feeding buzzes (6.8 % of total calls) at 47 sites during the study. Site activity levels (63.6 ± 31.7 average nightly passes for *M. norfolkensis* and 9.1 ± 3.3 average nightly passes for *M. species 2*) ranged from 0 - 1272.5 average nightly passes for *M. norfolkensis* and 0 – 144.5 average nightly passes for *M. species 2*. Lower foraging activity was recorded for *M. species 2* (0.31 ± 0.11, range...
0 - 4 feeding buzzes) than *M. norfolkensis* (2.14 ± 1.31, range 0 - 60 feeding buzzes). *Mormopterus* species 2 was more widespread, occurring at 72.3 % of sites compared to *M. norfolkensis*, which occurred at 57.4 % of sites. *Mormopterus* species 2 was recorded at each of the different habitat types sampled and *M. norfolkensis* was recorded at all habitat types except for the urban matrix (Figure 5-2). We recorded very high levels of activity of *M. norfolkensis* at mangrove forest sites (403.4 ± 170.7 average nightly passes) being over 50 times greater on average than other habitat types (freshwater wetland had the second highest activity with 7.9 ± 2.1 average nightly passes; Figure 5-2).

![Figure 5-2: Average nightly bat activity](image)

Average nightly bat activity (number of passes) recorded for each species in the study area across a) all seven landscape categories and b) landscape categories excluding mangrove forest. Abbreviations are: *Mormopterus norfolkensis* (Mono); *M. species 2* (Msp2); FWW - freshwater wetland; DSF - dry sclerophyll forest and woodland; DSFS - dry sclerophyll forest and woodland on sand; UBUSH - urban bushland remnant; UMAT - urban matrix; MAN - mangrove forest; and SOFF - swamp oak forest.
Percentage foraging activity differed significantly between freshwater wetland, mangrove forest, swamp oak forest and dry sclerophyll forest on sand habitat types for *M. norfolkensis* (df = 3, $\chi^2 = 25.653$, p < 0.001) and also between freshwater wetland, mangrove forest, and dry sclerophyll forest on sand habitats for *M. species 2* (df = 2, $\chi^2 = 33.868$, p < 0.001). The freshwater wetland habitat type had the greatest percentage foraging activity for both species (Figure 5-3). No foraging activity by either species was recorded in the urban habitat types and no *M. species 2* foraging activity was recorded in swamp oak forest.

![Figure 5-3: Average percentage foraging activity](image)

**Figure 5-3: Average percentage foraging activity**

Average percentage foraging activity recorded for *Mormopterus norfolkensis* (Mono) and *M. species 2* (Msp2) across seven different habitat types. FWW - freshwater wetland; DSF - dry sclerophyll forest and woodland; DSFS - dry sclerophyll forest and woodland on sand; UBUSH - urban bushland remnant; UMAT - urban matrix; MAN - mangrove forest; and SOFF - swamp oak forest.

### 5.4.1 Local-scale habitat models

Habitat type best predicted the activity levels of both *M. norfolkensis* and *M. species 2* in the best-fitting GLMs compared to other environmental variables sampled. *Mormopterus norfolkensis* was more likely to be active in freshwater wetland, swamp oak forest and dry sclerophyll forest on sand compared to dry sclerophyll forest (p < 0.001), in
the best fitting model (Table 5-1; Figure 5-4). Urban habitats were less likely to be used by *M. norfolkensis* than dry sclerophyll forest in the best-fitting model, but this was not significantly different (Table 5-1; Figure 5-4). There were two other models with strong support (within 2 AICc points of the best-fitting model) and four models in the 95% confidence set for *M. norfolkensis*. In the first supported model, *M. norfolkensis* was likely to be more active at low elevations and in areas without urban land-use within 500 m (Table 5-1). In the second supported model, *M. norfolkensis* was likely to be more active in areas with no urban land-use within 500 m and that had sandplain and floodplain soils compared to soils on undulating slopes (Table 5-1). ROC plots indicated that the best-fitting model had excellent discrimination ability (AUC = 0.92; Table 5-1) and explained a considerable portion of the variation in the data (Nagelkerke’s $R^2 = 0.78$). Other models had poorer discrimination ability (AUC < 0.85) and explained less variation in the data (Nagelkerke’s $R^2 < 0.70$; Table 5-1).

**Table 5-1: Summary of local-scale habitat use models**

Summary from habitat analysis of a) *Mormopterus norfolkensis* and b) *M. species 2* surrounding a large *M. norfolkensis* maternity roost. Total activity was modelled using generalised linear models with a negative binomial distribution. Data presented includes the coefficient estimates, Nagelkerke’s $R^2$ and area under the curve for the receiver operating characteristic (ROC) values for the best-fitting and supported (within 2 AICc) models.

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<tr>
<td>Variable(s)</td>
<td>Estimate</td>
<td>SE</td>
<td>Z</td>
<td>p</td>
<td>Log(L)</td>
<td>AICc</td>
<td>Wi</td>
<td>ΔAICc</td>
<td>ROC</td>
<td>R²</td>
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<tr>
<td><strong>Model 2</strong></td>
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</tr>
<tr>
<td>Intercept</td>
<td>-0.65</td>
<td>0.70</td>
<td>-0.93</td>
<td>0.353</td>
<td>-92.19</td>
<td>193.53</td>
<td>0.19</td>
<td>0.64</td>
<td>0.83</td>
<td>0.66</td>
</tr>
<tr>
<td>Low elevation</td>
<td>3.41</td>
<td>0.74</td>
<td>4.58</td>
<td>&lt;0.001*</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Presence of urban land-use within 500 m</td>
<td>-1.75</td>
<td>0.65</td>
<td>-2.69</td>
<td>0.007*</td>
<td></td>
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<tr>
<td><strong>Model 3</strong></td>
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</tr>
<tr>
<td>Intercept</td>
<td>-0.70</td>
<td>0.69</td>
<td>-1.02</td>
<td>0.307</td>
<td>-91.26</td>
<td>194.29</td>
<td>0.18</td>
<td>1.41</td>
<td>0.85</td>
<td>0.70</td>
</tr>
<tr>
<td>Presence of urban land-use within 500 m</td>
<td>-1.69</td>
<td>0.67</td>
<td>-2.53</td>
<td>0.011*</td>
<td></td>
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</tr>
<tr>
<td>Undslope versus Sandplain</td>
<td>3.29</td>
<td>0.82</td>
<td>4.00</td>
<td>&lt;0.001*</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Undslope versus Floodplain</td>
<td>3.60</td>
<td>0.77</td>
<td>4.66</td>
<td>&lt;0.001*</td>
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<tr>
<td><strong>b) Mormopterus species 2</strong></td>
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</tr>
<tr>
<td>Intercept</td>
<td>0.00</td>
<td>0.52</td>
<td>0</td>
<td>1.000</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>DSF versus FWW</td>
<td>3.16</td>
<td>0.63</td>
<td>4.99</td>
<td>&lt;0.001*</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>DSF versus SOFF</td>
<td>0.88</td>
<td>0.76</td>
<td>1.15</td>
<td>0.251</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>DSF versus DSFS</td>
<td>2.42</td>
<td>0.65</td>
<td>3.72</td>
<td>&lt;0.001*</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>DSF versus UBUSHE</td>
<td>1.03</td>
<td>0.76</td>
<td>1.36</td>
<td>0.173</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>DSF versus UMAT</td>
<td>-0.92</td>
<td>1.00</td>
<td>-0.92</td>
<td>0.359</td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>

* denotes significant coefficients at α = 0.05. FWW - freshwater wetland; DSF - dry sclerophyll forest and woodland; DSFS - dry sclerophyll forest and woodland on sand; UBUSHE - urban bushland remnant; UMAT - urban matrix; MAN - mangrove forest; and SOFF - swamp oak forest.

*Mormopterus* species 2 was more likely to occur in freshwater wetland and dry sclerophyll forest on sand compared to dry sclerophyll forest (p < 0.001; Table 5-1; Figure 5-4), in the best fitting model. There was no significant difference in the activity of *M. species 2* in swamp oak forest, urban bushland and urban matrix compared to dry sclerophyll forest habitat types (Table 5-1; Figure 5-4). However, there was a trend for *M. species 2* to be more active in swamp oak forest and urban bushland and less active in the urban matrix than the dry sclerophyll forest (Table 5-1; Figure 5-4). There was one other model in the
95% confidence set for *M.* species 2, but it did not have strong support. ROC plots indicated that the best-fitting model had fair discrimination ability (AUC = 0.76) and it explained considerable variation in the data (Nagelkerke’s $R^2 = 0.77$; Table 5-1). There was no evidence of spatial autocorrelation in the residuals for models in the 95% confidence set for either of the study species ($P > 0.564$).
**Figure 5-4: Habitat model response plots**

Partial conditional response graph of the relationship between the relative activity (number of passes) and habitat types included in the best-fitting negative binomial generalised linear models for *Mormopterus norfolkensis* and *Mormopterus* species 2. The grey shading represents 95% confidence intervals. Outliers are indicated by a number next to the point which specifies the value. FWW - freshwater wetland; DSF - dry sclerophyll forest and woodland; DSFS - dry sclerophyll forest and woodland on sand; UBUSH - urban bushland remnant; UMAT - urban matrix; URB - urban sites; MAN - mangrove forest; and SOFF - swamp oak forest.
5.4.2 Validation of regional-scale habitat models

Regional-scale model accuracy measures were calculated using a probability of occurrence threshold (above which we classify a presence and below which we classify an absence) of 0.34 for both species, as determined from the value of the intersection of sensitivity and specificity measures (see Appendix A). The regional-scale models had good discrimination ability to the local data for both species as measured by AUC (> 0.72; Table 5-2), while kappa and TSS accuracy measures were slight to fair (0.255 – 0.317; Table 5-2). Presence records occurred generally proportional to predictions, indicating a good level of model calibration for *M. norfolkensis* (Figure 5-5). *Mormopterus norfolkensis* records did tend to occur more often than predicted in areas mapped as having a low probability of occurrence (0 – 0.4 probability of occurrence; Figure 5-5). However, prevalence was not greatly under-estimated for *M. norfolkensis* overall (Figure 5-5). Figure 5-6 illustrates the predictive accuracy of the regional-scale mapping for *M. norfolkensis*, displaying false negatives in the north of the study area (which coincides with the dry sclerophyll forest on sand habitat type) and false positives in the low-lying parts of Newcastle city. *Mormopterus* species 2 was more prevalent than predicted by regional-scale models (Figure 5-5) and false negatives occurred in western parts of the study area, coinciding with dry sclerophyll forest habitat types (Figure 5-6).
Table 5-2: Predictive accuracy measures

Predictive accuracy of regional-scale models (Chapter 2) to local-scale data collected during this study. The confusion matrix was constructed using a probability threshold of 0.34 for both species as determined by the intersection of sensitivity and specificity values. AUC – area under the curve calculated from the receiver operating characteristic; TSS – true skill statistic; CCR – correct classification rate.

<table>
<thead>
<tr>
<th>Accuracy measure</th>
<th>Mormopterus norfolkensis</th>
<th>Mormopterus species 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>AUC</td>
<td>0.757</td>
<td>0.719</td>
</tr>
<tr>
<td>Kappa</td>
<td>0.313</td>
<td>0.255</td>
</tr>
<tr>
<td>TSS</td>
<td>0.317</td>
<td>0.292</td>
</tr>
<tr>
<td>CCR</td>
<td>0.660</td>
<td>0.660</td>
</tr>
<tr>
<td>Sensitivity</td>
<td>0.667</td>
<td>0.676</td>
</tr>
<tr>
<td>Specificity</td>
<td>0.65</td>
<td>0.615</td>
</tr>
<tr>
<td>Prevalence</td>
<td>0.574</td>
<td>0.723</td>
</tr>
</tbody>
</table>
Figure 5-5: Model calibration plots

Calibration plots of the proportion of observed occurrence (collected during this study), against the predicted values of regional-scale probability of occurrence mapping (Chapter 2) with 95% confidence intervals, for *Mormopterus norfolkensis* and *Mormopterus* species 2. Sample size in each bin is given by the numbers above. A close fit to the diagonal indicates good model calibration.
Figure 5-6: Accuracy visualisation maps

Regional-scale probability of occurrence maps (Chapter 2) overlaid with local-scale bat activity for *Mormopterus norfolkensis* and *Mormopterus* species 2 with activity levels from the local study overlaid. The probability of occurrence of each species from the regional-scale models was classified into three categories: low (0 – 0.33); moderate (0.33 – 0.66); and high (0.66 – 1) to aid visualisation of prediction errors. The sensitivity = specificity occurrence threshold (0.34) obtained from the local-scale study was used to define the lowest category.
5.5 Discussion

Few studies test the performance of habitat models against systematically collected and independent datasets (Manel et al. 2001, Araujo and Guisan 2006, Mouton et al. 2010). Our findings suggest that whilst the regional-scale habitat models are useful to describe habitat for these poorly understood species, they should be used cautiously at a local-scale, particularly when the consequences of errors are severe. Habitat type categories best predicted the activity of both species, compared to other environmental variables sampled, confirming previous findings that both species are more active in floodplains and that the threatened *M. norfolkensis* avoids urban areas. Additionally, these habitat type categories are easily communicated to land managers, illustrating that they are not only good summaries of environmental variation at a local-scale, but also practical tools for conservation.

5.5.1 Local-scale habitat models

Both species were recorded more frequently and site activity levels were higher than recorded during previous studies in both urban and forested areas (Hourigan et al. 2006, Lloyd et al. 2006, Basham et al. 2010, Threlfall et al. 2011). Activity levels were also considerably higher than those recorded for the regional-scale study, where mangroves were not sampled and freshwater wetlands were under-sampled (Chapter 2). This allowed habitat use to be modelled using activity levels, which are preferable to presence / absence data that may inflate the importance of some habitats for mobile insectivorous bats due to false positives. The activity of both *M. norfolkensis* and *M. species 2* was best explained by habitat type categories, which were based on vegetation communities that reflect environmental conditions such as soil type, climate, geology and topography. Habitat types may also incorporate other aspects such as vegetation structure, condition, disturbance history and the presence of important components such as hollow-bearing trees or rock outcrops, depending on the target species and the project aims. In this way,
habitat types are likely to better represent the environmental variation in a particular area, compared to the combination of a small number of raw variables (such as elevation, aspect, etc.). The classification of landscapes into different habitat types by experts is difficult at large spatial scales and has been previously found to offer little benefit over raw environmental variables (Pearce et al. 2001). However, we found that the creation of habitat type spatial layers can be efficiently compiled based on vegetation mapping combined with local expert knowledge for small study areas. Additionally, broad habitat type categories are easily communicated to conservation managers and authorities, which is beneficial compared to interpreting complex relationships with multiple raw variables.

*Mormopterus norfolkensis* and *M.* species 2 were more active in mangrove forest than any other habitat type sampled. This is most likely due to the abundant hollow resources in the Hunter Estuary mangrove forests, which have been found to be important roosting habitat for *M. norfolkensis* maternity colonies (Chapter 4). The percentage foraging activity in the mangrove forest was relatively low for *M. norfolkensis* and radio-tracking of bats indicated that they quickly move out of mangrove forests upon exiting roosts (Chapter 4). Additionally, very little *M. norfolkensis* activity has been recorded near hollow-depauperate mangrove forests elsewhere in NSW (Laegdsgaard et al. 2004, Gonsalves et al. 2012). As such, we suggest that mangrove forests within the study area represent roosting habitat for *M. norfolkensis* and that the foraging activity recorded during this study was opportunistic as bats exited and returned to roosts. Mangrove forests may not be as important for *M.* species 2 with less activity recorded and almost 45 times fewer *M.* species 2 individuals were captured during trapping within the Hunter Estuary mangrove forests (Chapter 4).

When the mangrove forest sites were excluded from analyses, we found that both *M. norfolkensis* and *M.* species 2 were more active in freshwater wetland habitats and
percentage foraging activity was also greatest in freshwater wetland for both species. Riparian habitats often support high levels of bat activity, which is likely to be due to high insect prey abundance (Racey and Swift 1985, Walsh and Harris 1996, Hayes 1997, Grindal et al. 1999, Flaquer et al. 2009). Additionally, more productive soils (as are often associated with floodplains) have also been found to be related to high bat activity levels (Law et al. 2011a, Threlfall et al. 2011). These productive low-lying floodplain areas were previously predicted to be high habitat quality by regional habitat mapping for *M. norfolkensis* and *M. species 2* (Chapter 2) and our study confirms the importance of these habitats.

Both *M. norfolkensis* and *M. species 2* were more active in dry sclerophyll forest on sand compared to the dry sclerophyll forest on low undulating hills. The Tomago sandbeds (where this habitat type occurs) have characteristics (sandy soils and low vegetation types) that are typically considered to be associated with nutrient-poor environments. This contrasts with the high activity of both *M. norfolkensis* and *M. species 2* recorded in productive floodplain areas of the study area. It is possible that the high activity is attributed to the presence of hollow-bearing trees that provide roosting habitat for these species. However, koalas (*Phascolarctos cinereus*) which have been linked to high fertility and clay soils in Victoria and Queensland (Januchowski et al. 2008, McAlpine et al. 2008), also prefer these sandy soils within the study area (Lunney et al. 1998, McAlpine et al. 2008). This occurrence of preferred koala habitat on the sandbeds has been attributed to the historical clearing of more suitable floodplain habitats (Knott et al. 1998, McAlpine et al. 2008). However, it is possible that the Tomago sandbeds are actually more productive than the sandy soil suggests, perhaps due to the high water table. Further investigation into the factors that drive productivity, particularly in relation to ground water would be valuable.
5.5.2 Differential habitat use

Whilst floodplain habitats were used by both morphologically similar species, there were also differences in habitat use between the two species. Most notably, *M. norfolkensis* activity was negatively associated with urban areas, whereas urban bushland was positively associated with *M. species 2* activity levels. *Mormopterus norfolkensis* was never recorded in the urban matrix and rarely in urban bushland remnants and this confirms previous assertions that *M. norfolkensis* avoids urban habitats (Chapter 2; Threlfall et al. 2011). Habitat mapping from regional-scale models indicated that urban habitats were only marginal habitat for *M. species 2* (Chapter 2). This is also consistent with our local study where *M. species 2* was recorded at only one urban matrix site and whilst it was recorded relatively frequently in urban bushland, activity was low and foraging was rarely recorded. *Mormopterus norfolkensis* may be particularly sensitive to disturbance associated with urbanisation such as noise and artificial lighting (in contrast with *M. species 2* which has been found to be positively associated with urban lighting; Threlfall 2011), but this requires explicit testing. Alternatively, these subtle differences in habitat use by *M. norfolkensis* and *M. species 2* may represent spatial segregation to either prevent competitive interactions, or in response to them (see Arlettaz 1999, Nicholls and Racey 2006b). Interspecific interactions are poorly understood for bat communities and further research into competition and niche partitioning is required.

5.5.3 Validation of regional-scale habitat models

The descriptions of habitat use from regional-scale models were consistent with those from local-scale models. Additionally, the discrimination ability of regional-scale mapping, as assessed by AUC was fair to good. However, threshold-based accuracy measures (kappa and TSS) indicated that the regional models performed poorly with both false positive and false negative occurrences for *M. norfolkensis* and *M. species 2* at a local-scale. Additionally, model calibration for *M. species 2* suggested a systematic bias in
which the regional-scale model underestimates species prevalence (Franklin 2009). The broad level of agreement between these models suggests that they are useful to describe general habitat for these poorly understood species. However, our findings also illustrate that regional-scale models should not be used without testing for fine-scale prediction of habitat use where high levels of certainty are required. This is particularly the case when the consequences of errors are severe, such as using regional-scale models to allocate land for development or conservation.

5.5.4 Management implications

The high level of use of floodplain areas and the avoidance of urban areas by *M. norfolkensis* suggest that its threatened conservation status is warranted. Floodplains are subject to high anthropogenic pressures from agriculture and urban development, with less than 30% of floodplain vegetation communities remaining in NSW (Keith 2004). Whilst wetland restoration has been found to have a positive effect on local bat communities elsewhere (Menzel et al. 2005), it is important to monitor how particular management actions such as draining, insect spraying and installation of tidal floodgates affect insect and bat communities and these factors would benefit from further investigation. The findings of both regional and local-scale habitat models suggest that conservation efforts should focus on conserving and restoring floodplains, the sensitive management of riparian areas on private land and limiting urbanisation in these areas.

5.6 Acknowledgements

We would like to thank the land managers that provided property access for this project. Trent Penman assisted with general statistical discussions, Leroy Gonsalves, Evan Pickett and two referees provided helpful comments on this manuscript.
### Appendix A: Candidate variables

**Table S1: Description of candidate variables used in the analyses**

Values are mean ± se (range) or frequency of ordinal values. A patch was defined as > 10 trees occurring together and patch sizes > 500 m² were mapped.

<table>
<thead>
<tr>
<th>Candidate Variable</th>
<th>Scale</th>
<th>Description / source</th>
<th>Source</th>
<th>Values (n = 47)</th>
</tr>
</thead>
<tbody>
<tr>
<td>HabType</td>
<td>Site</td>
<td>Original stratification unit: freshwater wetland (FWW); dry sclerophyll forest and woodland (DSF); dry sclerophyll forest and woodland on sand (DSFS); urban bushland remnant (UBUSH); urban matrix (UMAT); mangroves (MAN) and swamp oak forest (SOFF).</td>
<td>Vegetation layer, aerial photography, site field data</td>
<td>FWW = 9; DSF = 8; DSFS = 8; SOFF = 5; UBUSH = 5; UMAT = 5; MAN = 7; URB† = UMAT + UBUSH</td>
</tr>
<tr>
<td>Elev_low</td>
<td>Site</td>
<td>Elevation (m) converted to binary elevation categories for analyses (1 is &lt;= 10; 0 is &gt; 10)</td>
<td>DEM</td>
<td>19.0 ± 3.6 (0 – 95.2)</td>
</tr>
<tr>
<td>Slope</td>
<td>Site</td>
<td>Slope (degrees). Excluded from analyses due to little variation</td>
<td>DEM</td>
<td>1.7 ± 0.4 (0 – 15)</td>
</tr>
<tr>
<td>NearHBT</td>
<td>Site</td>
<td>Nearest hollow-bearing tree (m)</td>
<td>Field data</td>
<td>111.5 ± 26.0 (0 – 750)</td>
</tr>
<tr>
<td>NearPatch</td>
<td>Site</td>
<td>Distance to nearest woody vegetation patch (m)</td>
<td>Vegetation layer</td>
<td>149.7 ± 48.7 (0 – 1441.7)</td>
</tr>
<tr>
<td>NearUrb_close</td>
<td>Site</td>
<td>Distance to nearest urban land-use (m) converted to binary category where 1 is ≤ 500 m and 0 is &gt; 500 m.</td>
<td>Urban land-use layer</td>
<td>0 = 23</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1 = 24</td>
</tr>
<tr>
<td>NearRiv</td>
<td>Site</td>
<td>Distance to nearest major river (m). Log transformed for analysis</td>
<td>Aerial photography</td>
<td>2812.0 ± 377.2 (60.7 – 9082)</td>
</tr>
<tr>
<td>DistMang</td>
<td>Site</td>
<td>Distance to mangrove maternity roost patches (m)</td>
<td>GIS analysis</td>
<td>6256.4 ± 570.0 (0 – 12745.2)</td>
</tr>
<tr>
<td>Soil</td>
<td>Site</td>
<td>Broad soil landscape type adapted from (Mattei 1995). Categorical variable: undulating slope; floodplain; or sandplain</td>
<td>Soil landscapes (Mattei 1995)</td>
<td>Floodplain = 22; Sandplain = 10; Undslope = 15</td>
</tr>
<tr>
<td>Candidate Variable</td>
<td>Scale</td>
<td>Description / source</td>
<td>Source</td>
<td>Values (n = 47)</td>
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<td>--------------------------------------------------------------------------------------</td>
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</tr>
<tr>
<td>FlyWidth_Narrow</td>
<td>Site</td>
<td>Binary flyway width category, where 1 is ≤ 10 m. A flyway is an opening where vegetation is &lt; 1m in height.</td>
<td>Site field data</td>
<td>0 = 25</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1 = 22</td>
</tr>
<tr>
<td>%Veg_cat</td>
<td>Landscape (0.25, 0.5, 1, 2.5, 5km buffers)</td>
<td>Percent vegetation within buffer of sample site converted to ordinal categories for analyses were 1 is ≤ 10 %; 2 is ≤ 80 % and 3 is &gt; 80 %</td>
<td>Vegetation layer</td>
<td>0.25: 1 = 14; 2 = 13; 3 = 20</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td>0.5: 1 = 14; 2 = 19; 3 = 14</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1: 1 = 13; 2 = 27; 3 = 7</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2.5: 1 = 13; 2 = 34; 3 = 0</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>5: 1 = 1; 2 = 46; 3 = 0</td>
</tr>
<tr>
<td>PatDens_low</td>
<td>Landscape (0.25, 0.5, 1, 2.5, 5km buffers)</td>
<td>Patch density – number of patches per 100 ha within the buffer of sample site. Converted to a binary variable where 1 is ≤ 5 and 0 is &gt; 5</td>
<td>GIS analysis vegetation layer</td>
<td>0.25: 0 = 39; 1 = 8</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.5: 0 = 20; 1 = 27</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1: 0 = 17; 1 = 30</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2.5: 0 = 13; 1 = 34</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>5: 0 = 18; 1 = 29</td>
</tr>
<tr>
<td>MaxPat_cat</td>
<td>Landscape (0.25, 0.5, 1, 2.5, 5km buffers)</td>
<td>Maximum remnant vegetation patch size (ha) within buffer of sample site. Converted to ordinal categories where 1 is ≤ 100 ha; 2 is ≤ 500 ha and 3 is &gt; 500 ha.</td>
<td>GIS analysis vegetation layer</td>
<td>0.25: 1 = 26; 2 = 15; 3 = 6</td>
</tr>
<tr>
<td></td>
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<td></td>
<td></td>
<td>0.5: 1 = 23; 2 = 15; 3 = 9</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1: 1 = 20; 2 = 16; 3 = 11</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2.5: 1 = 11; 2 = 15; 3 = 21</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>5: 1 = 0; 2 = 12; 3 = 35</td>
</tr>
<tr>
<td>MinPat_low</td>
<td>Landscape (0.25, 0.5, 1, 2.5, 5km buffers)</td>
<td>Minimum remnant vegetation patch size (ha) within buffer of sample site. Converted to binary variable where 1 is ≤ 1 ha; 0 is &gt; 1 ha.</td>
<td>GIS analysis vegetation layer</td>
<td>0.25: 0 = 26; 1 = 21</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.5: 0 = 21; 1 = 26</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1: 0 = 5; 1 = 42</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2.5: 0 = 0; 1 = 47</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>5: 0 = 0; 1 = 47</td>
</tr>
<tr>
<td>Candidate Variable</td>
<td>Scale</td>
<td>Description / source</td>
<td>Source</td>
<td>Values (n = 47)</td>
</tr>
<tr>
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</table>
| AWMPat_cat         | Landscape (0.25, 0.5, 1, 2.5, 5km buffers) | Area weighted mean remnant vegetation patch size (ha) within buffer of sample site. Converted to ordinal categories where 1 is ≤ 50 ha; 2 is ≤ 250 ha and 3 is > 250 ha. | GIS analysis - vegetation layer | - 0.25: 1 = 25; 2 = 10; 3 = 12  
0.5: 1 = 23; 2 = 8; 3 = 16  
1: 1 = 20; 2 = 11; 3 = 16  
2.5: 1 = 14; 2 = 17; 3 = 16  
5: 1 = 0; 2 = 23; 3 = 24 |
| EdgDens_cat        | Landscape (0.25, 0.5, 1, 2.5, 5km buffers) | Density of edge habitat (km / 100 ha) within buffer of sample site. Derived from patch perimeters – does not include scattered trees. Converted to ordinal categories where 1 is ≤ 1 km per 100 ha; 2 is ≤ 5 km per 100 ha and 3 is > 5 km per 100 ha. | GIS analysis - vegetation layer | - 0.25: 1 = 14; 2 = 13; 3 = 20  
0.5: 1 = 13; 2 = 11; 3 = 23  
1: 1 = 10; 2 = 12; 3 = 25  
2.5: 1 = 4; 2 = 22; 3 = 21  
5: 1 = 1; 2 = 29; 3 = 17 |
| %CoreAr_pres       | Landscape (0.25, 0.5, 1, 2.5, 5km buffers) | Percent core area within buffer of sample site. Core area was calculated as the central portion of each remnant vegetation patch > 100 m from the edge. Converted to binary core area presence category where 1 is ≥ 5 %; 0 is < 5 %. | GIS analysis - vegetation layer | - 0.25: 0 = 25; 1 = 22  
0.5: 0 = 23; 1 = 24  
1: 0 = 22; 1 = 25  
2.5: 0 = 20; 1 = 27  
5: 0 = 15; 1 = 32 |
| MeanCore_cat       | Landscape (0.25, 0.5, 1, 2.5, 5km buffers) | Mean core area (ha) within buffer of sample site. Core area was calculated as the central portion of each remnant vegetation patch > 100 m from the edge. Converted to ordinal categories where 1 is 0 ha; 2 is ≤ 5 ha and 3 is > 5 ha. | GIS analysis - vegetation layer | - 0.25: 1 = 21; 2 = 20; 3 = 6  
0.5: 1 = 16; 2 = 26; 3 = 5  
1: 1 = 13; 2 = 27; 3 = 7  
2.5: 1 = 1; 2 = 41; 3 = 5  
5: 1 = 0; 2 = 47; 3 = 0 |
| CoreDens_cat       | Landscape (0.25, 0.5, 1, 2.5, 5km buffers) | Number of core areas of remnant vegetation patches per 100 ha within buffer of sample site. Converted to ordinal categories where 1 is ≤ 1 core area per 100 ha; 2 is ≤ 10 core areas per 100 ha and 3 is > 10 core areas per 100 ha. | GIS analysis - vegetation layer | - 0.25: 1 = 21; 2 = 1; 3 = 25  
0.5: 1 = 16; 2 = 5; 3 = 26  
1: 1 = 14; 2 = 4; 3 = 29  
2.5: 1 = 10; 2 = 17; 3 = 20 |
<table>
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<th>Candidate Variable</th>
<th>Scale</th>
<th>Description / source</th>
<th>Source</th>
<th>Values (n = 47)</th>
</tr>
</thead>
<tbody>
<tr>
<td>%CorePat_cat</td>
<td>Landscape (0.25, 0.5, 1, 5km buffers)</td>
<td>Average percent of a patch that is core area within buffer of sample site. Converted to ordinal categories where 1 is ≤ 10 %; 2 is ≤ 30 % and 3 is &gt; 30 %.</td>
<td>GIS analysis - vegetation layer</td>
<td>5: 1 = 1; 2 = 38; 3 = 8</td>
</tr>
<tr>
<td>%FWW_pres</td>
<td>Landscape (0.25, 0.5, 1, 2.5, 5km buffers)</td>
<td>Percent of freshwater wetland within buffer of sample site. Converted to a binary presence category where 1 is &gt; 5 % and 0 is ≤ 5 %.</td>
<td>GIS analysis - vegetation layer</td>
<td>- 0.25: 0 = 37; 1 = 10</td>
</tr>
<tr>
<td>%URB_pres</td>
<td>Landscape (0.25, 0.5, 1, 2.5, 5km buffers)</td>
<td>Percent of urban land-use within buffer of sample site. Converted to a binary presence category where 1 is &gt; 5 % and 0 is ≤ 5 %.</td>
<td>GIS analysis - vegetation layer</td>
<td>- 0.25: 0 = 37; 1 = 10</td>
</tr>
<tr>
<td>AvgMinTemp Site</td>
<td>Site</td>
<td>Average minimum nightly temperature (°C) recorded during the sampling period of each site.</td>
<td>nearest BOM station</td>
<td>19.3 ± 0.3 (15.3 – 23.1)</td>
</tr>
<tr>
<td>AvgAnnRain Site</td>
<td>Site</td>
<td>Average annual rainfall (mm)</td>
<td>GIS – average annual rainfall</td>
<td>1079.8 ± 6.5 (1009 – 1182)</td>
</tr>
</tbody>
</table>

† The UBUSH and UMAT habitat type categories were combined to form the URB category for the *M. norfolkensis* models as no *M. norfolkensis* activity was recorded at any UMAT site.
Appendix B: Probability threshold graphs

a)

![Graph a]

b)

![Graph b]

Figure S1: Probability threshold graphs

Effect of the probability of occurrence threshold on the sensitivity (dotted line), specificity (dashed line) and kappa (black line) accuracy measures for regional-scale models (Chapter 2) tested against independent local-scale data collected for Mormopterus norfolkensis and Mormopterus species 2.
Chapter 6

Home range and resource utilisation functions for *Mormopterus norfolkensis*, a threatened insectivorous bat, in South-eastern Australia

Anna McConville, Bradley Law, Michael Mahony and Mark Thompson
Chapter 6 Home range and resource utilisation functions for *Mormopterus norfolkensis*, a threatened insectivorous bat, in south-eastern Australia

6.1 Abstract

Studies that investigate the characteristics of individual home ranges are fundamental to understanding ecology and behaviour and may be particularly useful in managing threatened species at a local-scale. *Mormopterus norfolkensis* is a small threatened molossid bat species found in south-eastern Australia. Recent studies have investigated landscape- and local-scale habitat use via ultrasonic recorders. However, nightly movements at the individual-level have not been rigorously investigated. We explored the home ranges of a *M. norfolkensis* maternity colony by radio-tracking 12 lactating females from fixed telemetry stations and quantified bearing error using two different methods. We created utilisation distributions (UDs) for four bats using fixed kernel density estimators with a broad smoothing parameter to account for the low accuracy of location points. We then related the continuous UD surface of use to broad habitat types (based on vegetation community) using a multiple linear regression technique to obtain resource utilisation functions. We found that lactating female *M. norfolkensis* from the same colony had very large (4,099.5 ± 528.6 ha), overlapping home ranges and flew over a large freshwater wetland significantly more, but dry sclerophyll forests and urban areas significantly less, than cleared areas. The use of estuarine habitat types was variable among individuals and was most likely a result of the presence of maternity roosts in this habitat type. While limited in sample size, our results indicate that home range habitat use was consistent with predictions from modelling studies using acoustic sampling. We hope that our case
study will encourage other researchers to apply the resource utilisation function technique to bats.

6.2 Introduction

Information regarding the way that species use habitat is essential to the development of suitable conservation management strategies. However, our knowledge of some faunal groups is still rudimentary, even in developed countries which have established conservation legislation, such as Australia. Insectivorous bats are one of these poorly understood groups, as their nocturnal behaviour, small size and high mobility makes them difficult to study. The actual conservation status of many insectivorous bats in Australia is not well known (Armstrong 2011, Milne and Pavey 2011, Pennay et al. 2011) and further research into bat ecology is urgently required (Lunney et al. 2011a).

Landscape-scale studies of habitat selection provide coarse-grained information which is valuable for regional conservation planning (Osborne et al. 2001, Radford et al. 2005, Manning et al. 2006, Threlfall et al. 2011). However, studies that quantify the characteristics of individual home ranges, which encompass both the resource (i.e. food and shelter) and social requirements of individuals, are fundamental to understanding ecology and behaviour (Kernohan et al. 2001). Such studies are also useful in managing threatened species at the population-level. Radio-tracking is a powerful method to study the movements of animals (White and Garrot 1990) and the development of tiny transmitters has allowed its application to the study of insectivorous bats (Amelon et al. 2009). For example, the characteristics of areas within individual or colony ranges may be compared to those available, providing important information regarding habitat selection and guidance for conservation management (Adam et al. 1994, Bontadina et al. 2002,

Various statistical methods are available to researchers investigating habitat use within home ranges (Amelon et al. 2009). The use of point-based methods to explore habitat use has been questioned when location points have poor spatial accuracy, such as those derived from triangulation of multiple bearings to moving animals (Montgomery et al. 2011). Additionally, traditional methods that compare habitat use with availability, such as compositional analysis (Aebischer et al. 1993), assume that use is even across the entire home range despite this being rarely observed (Millspaugh et al. 2006). Methods that provide a projection of the relative occurrence of a species across a landscape incorporate more information and offer a powerful alternative (Manly et al. 2001). The utilisation distribution (UD) is a probability density function that converts location points to a 3D surface to represent an animal’s use of space, with greater UD heights indicating areas of greater use (Kernohan et al. 2001, Millspaugh et al. 2006). This continuous UD surface can then be related to various spatially-defined resources to provide a resource utilisation function (RUF; Marzluff et al. 2004). The RUF methodology uses multiple linear regression while accounting for the spatial autocorrelation that is inherent in UDs (Marzluff et al. 2004). RUFs have recently been used to explore habitat use of individuals and to draw population-level inferences (Rittenhouse et al. 2008, Donovan et al. 2011, Eads et al. 2011, Kertson et al. 2011).

_Mormopterus norfolkensis_ is a rarely captured insectivorous bat species found in southeastern Australia. It is currently is listed as vulnerable under the New South Wales (NSW) Threatened Species Conservation Act 1995 and as vulnerable C1 under the International Union for Conservation of Nature (IUCN) red list (IUCN 2009). Recent habitat models based on data collected using ultrasonic recorders, indicate that _M. norfolkensis_ uses low-
lying floodplains and coastal wetlands, but avoids urban areas (Chapters 2 and 5). Nightly foraging movements at the individual level have found a mosaic of vegetation types used, but observations so far have been limited (Chapter 3) and home range characteristics have not been previously described. The discovery of a large maternity population of \textit{M. norfolkensis} using two patches of mangroves as roosts (Chapter 4) provided an opportunity to investigate individual and colony-level habitat use.

The aim of this study was to investigate the nightly movements of a \textit{M. norfolkensis} maternity colony by radio-tracking individuals, creating UDs and relating home range use to habitat type via RUFs. We predicted that 1) \textit{M. norfolkensis} would use low-lying habitat types (freshwater wetland and estuarine) and avoid urban and forested areas, based on previous habitat modelling (Chapters 2 and 5); and that 2) lactating female bats from the same maternity colony would have overlapping home ranges.

6.3 Methods

6.3.1 Study area

The study was conducted in the Hunter Estuary, NSW, at the Port of Newcastle (32°55'36"S 151°46'44"E; Figure 6-1) in south-eastern Australia. The area experiences a warm temperate climate (average monthly temperatures 8.4 - 25.6 °C) and average annual rainfall of 1134 mm (Bureau of Meteorology 2012). Bats were captured in the western portion of what is now a 2600 ha landmass known as Kooragang Island. Areas surrounding the capture site have been highly modified by a history of agriculture, coal mining and port-associated industries. Heavy industrial development occurs in the south-eastern portion of Kooragang Island, and a major aluminium smelter occurs to the north-west. The non-industrial areas of Kooragang Island are dominated by mangroves and saltmarsh, with pasture and small areas of freshwater wetland also occurring. Hexham
Swamp, a large 2500 ha wetland dominated by freshwater wetland vegetation communities at the time of the study, occurs to the west. Since the study, floodgates have been opened to re-establish natural tidal flushing and more estuarine conditions in Hexham Swamp. Further west of Hexham Swamp are areas of undulating hills on nutrient-poor soil which support mostly young dry sclerophyll forest with few hollows. Relatively modern (< 20 years) and low density (up to 14 dwellings / ha) residential areas fringe the southern boundary of Hexham Swamp and are mostly cleared, with small linear vegetation remnants occurring along drainage lines.
Figure 6-1: Site location map

Trap site location (black circle), woody vegetation cover (light grey shading), mature mangrove forest (black shading), freshwater wetland (cross hatch), urban land-use (black stipple) and major waterbodies (dark grey shading).

6.3.2 Radio-tracking

We radio-tracked the nightly movements of 12 lactating female *M. norfolkensis* during one maternity season (November and December 2010), with up to four individuals tracked at a time. We captured bats using harp traps (Austbat P/L, Victoria, Australia) set in mangrove forests. Lactating females were identified by the presence of bare patches around enlarged nipples or expression of milk. We attached radio-transmitters to the dorsal
surface of bats in between the scapulae using adhesive glue (VetBond, 3M). Two types of transmitter were used, five LB-2N (Holohil, Carp, Canada) with 14 cm aerials, referred to as ‘transmitter H’ and seven LT6-337 (Titley Electronics, Balina, Australia) with 15 cm aerials, referred to as ‘transmitter T’. Transmitters weighed between 0.35 g (Holohil) and 0.4 g (Titley Electronics), which represented ≤ 5 % of the body weight of individuals.

We located bats using fixed telemetry stations and from mobile observers in vehicles or on foot. We used receivers (Australis 26k and Regal 2000 from Titley Electronics, Ballina, Australia; TR2-Scanner from Telonics, Arizona, USA) and three element yagi antennae (Titley Electronics, Ballina NSW) to locate radio-tagged bats. Since bats have been found to be more active in the hours immediately after dusk (Law et al. 1998) and to minimise temporal variation in activity by pooling activity from different periods of the night (Rettie and McLoughlin 1999), bats were tracked 5 - 6 hours following dusk each night. We trialled tracking bats at night during a related study of roost selection (Chapter 4) using the close approach method (Amelon et al. 2009). However, we could not be confident where bats were located due to the inaccessible open terrain (wetland) and possibly the speed and height of bat flight. For example, on one occasion an observer recorded a tagged bat in very close proximity (a strong signal and estimated at < 100 m), but another observer also detected a strong signal from 1 km away indicating that the bat was actually somewhere in between. To minimise the subjective estimation of bat distance from observer using the close approach method, we chose to triangulate bat locations. Teams of two people were positioned at fixed stations that were located in elevated locations. One person operated the receiver and dictated the compass bearings and the other transcribed the details to field sheets and kept time. At each fixed station we installed a metal stake (approximately 1.2 m tall) into the ground. Over this we placed a plastic pipe tower (2.2 m) with an antenna and a compass attached (Appendix A). Care was taken to ensure the metal stake (or other objects) did not interfere with compass operation. The
plastic pipe of the tower revolved over the metal stake and this reduced operator fatigue
and aimed to standardise observations at the fixed stations. Mobile observers moved
around the study area in vehicles and on foot and obtained compass bearings to bats
using antennae and receivers (without the aid of the plastic tower). Communication was
maintained between teams using two-way radios and mobile telephones. Four teams
were usually active at any one time, three at fixed stations and one mobile team.

We used a systematic sampling approach and the null to null method (Amelon et al. 2009)
to obtain bearings to radio-tagged bats. We recorded the start and end (null) bearings of
bat signals using a compass every 2 min using synchronised watches and on a rotating
basis, resulting in 8 min between bearings recorded for each tracked bat. Notes on signal
strength, consistency and receiver settings (gain and volume) were taken to aid in the
exclusion of erroneous bearings during the analysis phase. Mobile teams used the
scanning function of receivers to continually search for bats and recorded bearings to any
signals found synchronously with the fixed stations. The centre point of the start and end
(null) bearings was later calculated and converted to grid north to be used as the bearing
in the analyses.

The positions of fixed stations were later recorded to within 50 mm using a differential
global positioning system (GPS; Sokkia GSR2700 ISX). Other locations from which
bearings were taken by mobile observers were recorded using a standard hand-held GPS
(eXplorist 300, Magellan) which are usually accurate to within 10 m. Day roosts were also
located for a concurrent study on roost selection using the hand-held GPS (Chapter 4).

6.3.3 Bearing error tests

During the study, we used a transmitter lost in a tree (the ‘fixed transmitter’), to calculate
the bearing error associated with bearings taken (n = 95) by observers at a fixed station
over four nights (22 - 25 October 2012) using the protocol outlined for fixed stations above. We used a flat linear track in Hexham Swamp (where the bats were frequently recorded) to perform a ‘linear distance test’ at night on 2 July 2012. Two transmitters were placed at increasing distances (250 m, 500 m, 750 m, 1000 m, 1500 m and 2000 m) from the receiver; Appendix B) on timber poles at a height of 5.1 m to simulate bats in flight (Appendix B). We ensured that transmitters were separated by > 200 m during tests to reduce the potential for interference between transmitters. We used a plastic pipe tower in the same manner as described for fixed telemetry stations above with three telemetry receiver units (Regal 2000, Titley Electronics, Balina, Australia) of varying age. During the linear distance test, the operator took three sequential bearing measurements from each receiver unit to each transmitter at each distance, resulting in 108 attempted bearings (attempts to locate transmitters at the farthest distance categories were sometimes unsuccessful). The order that the receiver units were used was randomised and the observer moved the antennae to a wide start position before taking each bearing.

The location of the transmitters and fixed station used in the linear distance test were determined using a differential GPS to within 200 mm and 50 mm, respectively, to allow true bearings to be calculated with high accuracy. The bearing error (degrees) was calculated as the positive difference between the observed and true bearings to each transmitter. We used a one-way analysis of variance (ANOVA) to test for effects of distance on bearing error (log₁₀ transformed) and we examined post-hoc differences using Tukey’s HSD test. Additionally, the effects of receiver unit on bearing error were examined using a t test.

6.3.4 Utilisation distributions

We used Locate III (Nams 2006) to triangulate bat locations and calculate 95 % confidence error ellipses (using the standard deviation of the bearing error calculated
during the linear distance test; 8.3 °). We inspected triangulated locations and excluded 29 locations with error ellipses that were > 1,000 ha in size. UDs were calculated for each individual bat with ≥ 18 location points (four bats) using a fixed kernel density estimator in R (R Development Core Team 2008) using the ‘adehabitatHR’ package (Calenge 2006).

We first selected the smoothing parameter (h) for individual UDs using the least squares cross validation (LSCV) method as recommended by Seaman et al. (1999). However, this resulted in home ranges that had multiple small centres of activity for two bats. As the error ellipses associated with locations were relatively large (Appendix C), small centres of activity placed more certainty on locations than is appropriate. So for the final UDs we selected a h value based on visual inspection of the home range sizes (Brearley et al. 2011) within the minimum and maximum h value range determined by LSCV and used the same value for all bats. We viewed UDs created with increasing h values and selected one value that was slightly greater than when the small centres of activity first merged together for all bats (h = 800). Previous investigations have suggested that while h influences the extent and ruggedness of the UD, it does not grossly affect the determination of relative use or the relationship between relative use and resource occurrence (Kertson and Marzluff 2011). We included roost locations for each individual in the location data. However, we deleted roosts that were within 50 m of each other and only used a roost location once, even if it was used by the bat on multiple nights (Seaman et al. 1998).

To determine the minimum sample size required to estimate home ranges, we randomly sampled locations and then calculated the home range size within the 95 % volume contour for every additional locational point for 20 repetitions, starting at five locations (Appendix D). We determined if there were sufficient locations recorded if the graph reached an asymptote, which we defined as the point where three or more additional
locations resulted in equivalent or decreased home range size. The average number of locations in which an asymptote was reached was $19 \pm 5.0$ and one bat (M341) did not reach asymptote (Appendix D). However, we opted to include M341 in further analyses to explore use but have considered this throughout our discussion.

In addition to the UDs of the four individual bats that had $\geq 18$ location points, we also created a UD ($h = 800$) using pooled locations from all 12 bats tracked and used the 99.9 % volume contour, to define the area of habitat available to individuals (the ‘colony range’). The individual and colony UDs were calculated using the same resolution (25 m) and extent and we excluded cells outside of the 99.9 % volume contour of the colony range using Hawth’s Tools (Beyer 2007) in ArcGIS (version 9.3, ESRI, Redlands, CA, USA).

6.3.5 Home range size and overlap

We used the kerneloverlaphr() function of the adehabitatHR package (with 95 % volume contour setting) to calculate home range overlap measures from the individual bat UDs, including the percentage overlap, volume of intersection statistic (VI), the Bhattacharyya’s affinity (BA) and the utilisation distribution overlap index (UDOI; Fieberg and Kochanny 2005). The VI and BA values range from 0, which indicates no overlap between two home ranges and values of 1 for identical home ranges. The UDOI also equals 0 when there is no home range overlap and equals 1 if both UDs are uniformly distributed and have 100 % overlap. However, UDOI can be $> 1$ if the UDs are non-uniformly distributed and have a high degree of overlap. UDOI values $< 1$ indicate UDs that have less overlap and are non-uniformly distributed (Fieberg and Kochanny 2005).
6.3.6 Habitat use - used and available

We first explored habitat use by comparing the proportion of each major habitat type used by each individual to the proportion of habitat available to the colony. To do this, we created a map (25 m resolution) in ArcGIS of the five broad habitat types (based on vegetation communities) that existed within the colony range: estuarine (mangrove forest, saltmarsh, swamp oak forest and the Hunter River), cleared (mostly grassland on productive alluvial flats used for cattle grazing), freshwater wetland (reeds and open water), dry sclerophyll forest (also includes small areas of wet sclerophyll forest) and urban (suburban and industrial land-uses with little native vegetation). We then calculated the area of each habitat type within individual home ranges (used habitat) and within the colony range (available habitat) and expressed each area as a percentage of the home or colony range which was then visualised to explore trends in habitat use.

6.3.7 Habitat use - resource utilisation functions

Secondly, we created RUFs to relate the UD surface of each individual to the different habitat types within the colony range. As we created the individual UDd with an extent defined by the colony range, the RUFs relate individual use to habitat types across the entire colony range (second-order selection; Johnson 1980). We used the getvolumeUD() function in the adehabitatHR package in R to convert the UD from density to percent volume, so that the value of each pixel is equal to the percentage of the smallest home range containing this pixel. For example, a pixel located on the outskirts of a home range, such as under the 99 % volume contour, would have a value of 99. To minimise confusion during interpretation, we inverted the percent volume of each pixel, so that ‘use’ (the response variable in subsequent analyses) ranged from pixel values of 0 (lowest use) to 100 (highest use) across the UD surface (following Kertson et al. 2011). We then clipped the habitat map (described above) to the colony 99.9 % volume contour home range using
Hawth’s Tools in ArcGIS, imported the map into R and extracted the habitat type and use value for each pixel into a table for each individual.

To create RUFs for individual bats, we used the unstandardised ruf.fit() function of the package ‘ruf’ (Marzluff et al. 2004, Handcock 2012) in R with a spatial range value of 800 (the $h$ value used in the kernel estimate) and smoothness value of 1.5. Use by each individual was considered to be significantly associated with a particular habitat type if the 95 % confidence intervals for that habitat type did not overlap with the confidence intervals of the reference category (the cleared habitat type) via visual assessment (following Cumming et al. 2007). To explore consistency in habitat use among individuals, we followed Marzluff et al. (2004) in creating a ‘population RUF’. To do this, we averaged betas from individual RUFs and calculated standard errors and 95 % confidence intervals using a variance estimate that accounted for inter-animal variation (Eq3 in Marzluff et al. 2004). As we had only four individuals, the population RUF, may not actually represent habitat use by the population. However, it is valuable to explore the amount of variation among individuals and we use the term ‘population RUF’ for consistency with other publications. For population RUFs, we tested for statistical differences between coefficients ($\alpha = 0.05$) using paired t tests. All results are reported as mean ± standard error and the absolute value of RUF betas, unless otherwise specified.

### 6.4 Results

A total of 1256 bearings were recorded and 171 bat locations were calculated from triangulated or bisected bearings to 12 bats, with a median error ellipse area of 212.2 ha (range 1.2 - 983.9 ha; Appendix C). Bats were tracked for 4.5 ± 0.5 nights on average (range 2 - 7) and the four bats for which we calculated home ranges (M341, M342, M344 and M345) we recorded an average of 35.1 ± 6.0 bearings (range 0 - 102) and 5.1 ± 1.4
locations (range 0 - 24) each night. Observations of roosts at dusk found that bats moved quickly out of signal range after exiting roosts (average 6.5 ± 1.3 mins; n = 10, range 2 - 15 mins; Chapter 4). Bats appeared to fly directly from mangrove roosts towards Hexham Swamp, the large freshwater wetland. Additionally, later in the night (11pm - 12am), we sometimes recorded fast, direct flight towards the mangroves, which may indicate bats returning to roosts to feed young.

6.4.1 Bearing error tests

The average bearing error recorded from the linear distance test was 8.2 ± 1.0 ° (SD 8.5 °, range 0.4 - 39.1 °) and this was similar to the bearing error calculated to the fixed transmitter during the radio-tracking study (8.7 ± 0.8 °, SD 8.2 °, range 0.2 - 47.2 °; Appendix E). Overall, 80 bearings (74 %) were successful during the linear distance test, with transmitter H received at a greater distance (1500 m) than transmitter T (1000 m). Bearing error was significantly different between distance categories (F 4, 75  = 5.538, p = 0.006; Appendix B). Bearing error was two to three times greater at the 250 m and 1500 m locations (16.2 ± 2.8 ° and 11.9 ° ± 2.9 respectively) than the mid-distance categories of 500 m, 750 m and 1000 m (5.4 ± 0.8 °, 4.7 ± 1.1 ° and 5.0 ± 1.2 ° respectively; Appendix B). Post-hoc tests found that the close 250m distance category was significantly different from the mid-distance categories (500 m, 750 m and 1000 m; p < 0.033), but other distance category comparisons were non-significant. The receiver unit did not have a significant effect on bearing error (F2, 77 = 0.372, p = 0.691).

6.4.2 Home range characteristics

We calculated home ranges for four bats that had ≥ 18 locations and found that the average home range size (within the 95 % volume contour) was 4,099.5 ± 528.6 ha (Figure 6-2; range 2,525.7 – 4,806.3 ha) and the colony range (within the 99.9 % volume contour) was 11,151.4 ha (Figure 6-2). The distance from one side of the home range to
the other at the widest point was 10.3 ± 1.0 km (range 7.6 - 11.9 km). M341, M342 and M344 were tracked simultaneously and the UDs and home ranges of M341 and M342 had the highest levels of overlap (Table 6-1; Table 6-2; Figure 6-2). The home range of M344 was the smallest and was nested within the home ranges of M341 and M342 (Figure 6-2; percent overlap of 92.7 % and 96.1 % respectively; Table 6-1). The UD and home range of M345 overlapped least with the other three bats tracked, but still overlapped somewhat (Table 6-1; Table 6-2; Figure 6-2).

Figure 6-2: Home ranges of four lactating female Mormopterus norfolkensis

Home ranges (black lines) of four lactating female M. norfolkensis, derived from the 95 % volume contour (fixed kernel; \( h = 800 \)), shown over habitat type categories within the colony home range which was derived from the 99.9% volume contour (fixed kernel; \( h = 800 \)) calculated from pooled locations from all bats radio-tracked (location \( n = 156 \)). The habitat type categories are: estuarine (light grey shading); freshwater wetland
(crosshatch shading); dry sclerophyll forest (dark grey shading); urban (black stipple) and cleared (white) within the colony home range. Approximate trap location is illustrated by the black triangle.
Table 6-1: Percentage home range overlap

Percentage overlap of individual home ranges calculated using the 95 % fixed kernel volume contour. The percentage of the home range of bats in the first column that occurs within the home range of each bat in the top row is calculated.

<table>
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<th>M341</th>
<th>M342</th>
<th>M344</th>
<th>M345</th>
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<td>96.1</td>
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<td>69.4</td>
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<td>59.0</td>
<td>66.5</td>
<td>38.8</td>
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Table 6-2: Utilisation distribution home range overlap measures

Home range overlap measures (Fieberg and Kochanny 2005) calculated within the home range (95 % volume contour) to describe the degree of similarity among utilisation distributions (UDs). The volume of intersection statistic (VI) and the Bhattacharyya’s affinity (BA) values range from 0, which indicates no overlap between two home ranges and values of 1 for identical home ranges. The utilisation distribution overlap index (UDOI) also equals 0 when there is no home range overlap and equals 1 if both UDs are uniformly distributed and have 100% overlap. However, UDOI can be > 1 if the UDs are non-uniformly distributed and have a high degree of overlap. Values < 1 indicate UDs that have less overlap and are non-uniformly distributed.

<table>
<thead>
<tr>
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<th>Method</th>
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<th>M342</th>
<th>M344</th>
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</thead>
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<tr>
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<td>VI</td>
<td>0.696</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>BA</td>
<td>0.904</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>UDOI</td>
<td>1.032</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>M344</td>
<td>VI</td>
<td>0.613</td>
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<td>BA</td>
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<td>0.923</td>
<td>-</td>
</tr>
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<td>UDOI</td>
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<td>UDOI</td>
<td>0.333</td>
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</table>
6.4.3 Habitat use

On average, the four *M. norfolkensis* lactating females used freshwater wetland 20.7 ± 3.6 % more, dry sclerophyll forest 13.3 ± 1.9 % less and urban 6.1 ± 0.9 % less than expected based on availability to the colony (Figure 6-3). The use of cleared and estuarine habitat types (mean difference -1.9 ± 1.8 % and 0.6 ± 2.5 %, respectively) was proportional to the availability of these habitat types (Figure 6-3).

![Figure 6-3: Average difference between used and available habitat types](image)

The population RUF indicated that together, lactating female *M. norfolkensis* used freshwater wetland significantly more than cleared habitat types (*t*₃ = -3.089, *p* = 0.054;
In addition, use was significantly less in dry sclerophyll forest ($t_3 = 3.572, p = 0.038$) and urban ($t_3 = 4.252, p = 0.024$) habitat types when compared with cleared areas (Figure 6-4; Figure 6-5). The use of estuarine habitat types was not significantly different from the use of cleared habitat types in the population RUF ($t_3 = -0.504, p = 0.649$; Figure 6-5), as use varied between individuals. M341 and M342 used the estuarine habitat type significantly less than cleared, whereas estuarine habitat use was not significantly different from cleared for M344 and M345 (significance estimated visually; Figure 6-4). See Appendix F for more RUF details.

Figure 6-4: Individual resource utilisation function response plots

Response plots of use from resource utilisation functions of four lactating female *M. norfolkensis*, with 95 % confidence intervals. Use values are the inverted percent volume contour derived from utilisation distributions
created using fixed kernel methods, where a value of 0 indicates low use and a value of 100 indicates very high use within the colony range. Individuals that have use significantly associated with a habitat type, have 95% confidence intervals that do not cross the confidence intervals of the reference category (cleared). Significant values below the reference category (cleared) indicate a negative association and values above indicate a positive association with that resource. The habitat type categories were CLE – cleared; DSF – dry sclerophyll forest; EST – estuarine; FWW – freshwater wetland; and URB – urban.

**Figure 6-5: Population resource utilisation function response plot**

Response plot of use from population resource utilisation function (derived by averaging coefficients from four lactating female *M. norfolkensis*), with 95% confidence intervals that take into account inter-animal variation. Use values are the inverted percent volume contour derived from utilisation distributions created using fixed kernel methods, where a value of 0 indicates low use and a value of 100 indicates very high use within the colony range. * denotes significance at $\alpha = 0.05$ via paired $t$ tests with the reference category (cleared). The habitat type categories were CLE – cleared; DSF – dry sclerophyll forest; EST – estuarine; FWW – freshwater wetland; and URB – urban.

Within the overlapping home ranges, the freshwater wetland habitat type was mostly Hexham Swamp, which was characterised by dense reeds at the time of the study, with only small areas of open water. The cleared habitats were mainly low-lying paddocks on
productive alluvial flood plains, with scattered trees and small remnants (mostly *Casuarina glauca* swamp oak) used for cattle grazing. Urban areas that were included within home ranges consisted mostly of modern residential estates on the edge of Hexham Swamp. Some of our fixed telemetry stations were located between Hexham Swamp and the residential estates and only once was a very strong signal (indicative that the bat was very close) recorded. Estuarine habitat types within home ranges consisted of the two mangrove patches in which roosts were located, regenerating saltmarsh areas on Kooragang Island and the Hunter River.

6.5 Discussion

This is the first study to investigate home ranges of an Australian molossid and it provides important information regarding habitat use surrounding a maternity colony. Consistent with our predictions, we found that the four lactating female *M. norfolkensis* from the same colony had large, overlapping home ranges and flew over freshwater wetland significantly more, but dry sclerophyll forest and urban areas significantly less, than cleared areas. Our study when combined with a study of roost selection (Chapter 4), tentatively suggests that the Hunter Estuary *M. norfolkensis* maternity colony requires a mosaic of different habitat types, preferring freshwater wetland for foraging and forests with a high density of hollow-bearing trees (mature mangrove forests) for roosting. We hope that our case study will encourage other researchers to apply UDIs RUFs to bats.

6.5.1 Limitations and bearing error

Vegetation and other obstacles restrict the transmission of signals and as such the detection rate of animals in radio-telemetry studies can vary depending on habitat type (White and Garrot 1990, Amelon et al. 2009). We established fixed telemetry stations in elevated locations based on the flight path of bats, with the low-lying Hexham Swamp
used regularly by bats. The narrow and linear nature of the mangrove forests meant that bats were readily detected from outside of forests once they had left roosts and as such it is unlikely that bats were undetected in mangrove forests close to their capture site. However, inaccessibility to the field researchers of more distant mangrove forests to the north-east of the capture site meant that these areas were not well covered. However, bats did not appear to be moving towards these inaccessible areas prior to signal loss. Mobile observers regularly searched other habitat types, such as dry sclerophyll forests for missing bats and fixed stations that were close to forests rarely recorded bats moving towards forests, especially prior to reception being lost. Therefore, we consider that the effect of different detection rates on our analyses was minor.

Bats were also unable to be located at all times and it is possible that they were using areas outside of our modelled home ranges. Mobile observers regularly searched away from Hexham Swamp for missing bats, usually in increasing distances and in all directions from roosts and as such it is unlikely that bats regularly moved to other areas without detection. Many of the triangulated locations of bats were at greater distances from observers than we could detect transmitters during the bearing error tests (1 - 1.5 km) and it is likely that bats were flying higher than our test transmitters (> 5.1 m), as reception range increases with height (White and Garrot 1990). If bats were flying slightly lower at other times, they may not have been able to be detected over the wide wetland (which we were unable to access). Additionally, remnant vegetation patches may have obscured signals in some areas.

We found that the bearing error associated with locating bats was within the results range obtained from other similar studies (< 15 ° in Dechmann, Kranstauber et al. 2010; 9.3 ° SD in Bontadina, Schofield et al. 2002; and 4 ° in Adam, Lacki et al. 1994). Observers, equipment and the study area (adjacent to a major urban centre with various sources of
signal interference) may all have contributed to the locational errors that we have reported. In order to address this, we rigorously quantified bearing error to provide context for interpretation of the results and created UDs using fixed kernel methods with a large smoothing parameter to reflect the inaccurate nature of the location points. Additionally, to minimise the effect of telemetry error on our results, we specifically avoided using location points as response variables in analyses and used habitat types that were restricted to coarse-grain categories.

Highly mobile animals are difficult to radio-track and despite substantial effort we were able to triangulate relatively few locations. As a result, UDs were created for only four (33 %) of the animals tracked and with relatively few location points (Seaman et al. 1999 recommend > 30 location points are used). As we tracked bats for the first half of the night only, it is also important to note that our UDs may not represent the entire home range of individuals. However, we have considered the limitations associated with our small sample size and therefore, restrict our conclusions regarding population-scale habitat use and specifically discuss limitations where appropriate. Our findings are useful to explore habitat use for a threatened and rarely captured species and provide a basis for further population-scale research. Research aimed at further developing techniques that incorporate telemetry error, such as those that incorporate location error ellipse characteristics (size and spatial location), would be valuable.

6.5.2 Home range characteristics

The maximum distance across *M. norfolkensis* home ranges, approximated the 11 km travelled from roost to foraging grounds by a much larger molossid bat (30 - 40 g), *Tadarida australis* in northern Australia (Rhodes and Catterall 2008). Home ranges in our study were large compared to other molossid bats. For example, *Chaerephon pumilus* and *Mops condylurus* in Africa had average home range sizes of 1,319 ha and 1,437 ha,
respectively (Noer et al. 2012). Studies of lactating females of moderately mobile species elsewhere recorded much smaller home ranges, such as a 350 ha (median) for *Chalinolobus tuberculatus* in New Zealand (O'Donnell 2001), 7.6 ha (average) for *Myotis lucifugus* in Canada (Henry et al. 2002) and 304 ha (average) for *Myotis volans* in North America (Johnson et al. 2007). The formation of large home ranges, such as we recorded, has been proposed as a response to a sparse distribution of insect prey (O'Donnell 2001). However, riparian areas (like our freshwater wetland vegetation type) generally have abundant insect prey (e.g. Racey and Swift 1985, Fukui et al. 2006) and as such, it is unlikely that prey scarcity is driving the large home range sizes we recorded over freshwater wetland. It is possible that the location of the maternity roosts in the locally unique mangrove forest (Chapter 4) required lactating females to travel further to foraging grounds and thus have larger home ranges (Popa-Lisseanu et al. 2009). Indeed, the activity of *M. norfolkensis* in the mangrove forest decreases outside of the maternity season (A. McConville unpublished data), indicating that mangrove roosts may not be as heavily used at other times of year. Home ranges are unlikely to be static over time (De Jong 1994) and may vary in response to prey abundance or reproductive status. Therefore, it is also possible that home range sizes decrease when females are not lactating (Popa-Lisseanu et al. 2009) or that home ranges are large a result of spacing to minimise interspecies interactions such as competition for foraging resources (O'Donnell 2001).

The individuals tracked had moderately high levels of home range overlap, indicating at a preliminary level, that the maternity colony shares foraging resources (particularly for M341, M342 and M344 which were tracked concurrently). However, individuals were not simultaneously recorded in the same direction and it appears unlikely that group foraging occurred as has been found in another small molossid species, *Molossus molossus*, in Panama (Dechmann et al. 2010). Despite this, some level of communication regarding
foraging grounds may occur between members of the maternity colony and further investigation of social interaction is required.

6.5.3 Habitat use

Our results are mostly consistent with predictions from previous habitat modelling studies of *M. norfolkensis* in the Hunter Estuary (Chapters 2 and 5), with freshwater wetland being highly important and the urban and dry sclerophyll forest habitat types avoided. Similarly, *T. australis* was also found to prefer floodplain habitats in northern Australia, but was less discriminating in its use of urban areas (Rhodes and Catterall 2008). Globally, riparian areas have been found to be important to insectivorous bats (Racey and Swift 1985, Grindal et al. 1999, Holloway and Barclay 2000, Monadjem and Reside 2008, Popa-Lisseanu et al. 2009). High insect abundance in riparian areas have been found to be correlated with high insectivorous bat activity (Racey and Swift 1985, Fukui et al. 2006) and particularly, the emergence of aquatic insects has been found to be one of the most important factors influencing the distribution of bats in riparian areas (Fukui et al. 2006).

However, in slight contrast to prediction 1, the relative use of estuarine habitat types varied among individuals. Two patches of mangrove forest (approximately 15 ha and 8 ha in size and 10 - 15 m tall) with a high density of hollow-bearing trees were used almost exclusively for roosting by the maternity colony (Chapter 4) and bats were observed to move quickly from the mangrove forest after exiting roosts. While a previous study relying on ultrasonic detectors (Chapter 5) recorded high *M. norfolkensis* activity levels within mangrove forests, our radio-tracking study suggests that freshwater wetland is a far more important foraging resource for maternity colonies and high activity recorded previously in the mangrove forests is likely to reflect the numerous roosts present.
6.5.4 Management implications

All four lactating females were significantly associated with the freshwater wetland communities of Hexham Swamp and three out of four bats used Hexham Swamp above all other available habitat types. At the time of the study, three floodgates closed in the 1970’s to limit the natural tidal inflow (and thus create freshwater conditions rather than saline) had been gradually re-opened. Following the study, in late 2011, an additional three floodgates were opened and saline vegetation communities were re-establishing in the eastern portion of the wetland. Such alterations to the hydrology of the wetland, even though returning to a natural state which is desirable, are likely to alter aerial invertebrate diversity and abundance. How these changes have or will influence the insectivorous bat community assemblage and in particular, the *M. norfolkensis* maternity colony (whether positive or negative effect), requires further investigation. The Hunter Estuary contains high quality remnants of saline wetlands, such as saltmarsh and mangroves, in close proximity to *M. norfolkensis* maternity roosts. However, the use of estuarine habitats by lactating females was variable, whereas the use of freshwater wetland was consistent, suggesting that freshwater, rather than saline, wetlands are preferred. If this is correct, we would expect *M. norfolkensis* foraging habitat use would contract to western areas of Hexham Swamp which will retain freshwater wetland ecosystems. If the maternity colony remains roosting in the mangrove forest, this may place further pressure on lactating females during this energetically demanding time.

Studies of *M. norfolkensis* maternity colonies in other regions are required to test whether landscapes which have a mosaic of different habitat types (see Law and Dickman 1998), particularly productive floodplains in close proximity (within 5 km) to forests with abundant hollow-bearing trees are most important to lactating *M. norfolkensis*, as tentatively suggested by our preliminary results. At a local-scale, the restoration of highly disturbed floodplain wetlands, such as those historically converted to pasture, may also benefit
M. norfolkensis, particularly where roosting habitat is nearby. The protection of hollow-bearing trees within 5 km of floodplains and particularly, remnant patches with a high abundance of hollows should be undertaken as a conservation priority. Finally, for development proposals in these important floodplain areas, targeted surveys to identify M. norfolkensis maternity roosts should be undertaken to ensure that impacts on these critical habitat features are adequately considered.

6.6 Acknowledgements

We would like to thank the numerous dedicated volunteers (too many to list here) that assisted with the field work for this project and land managers that provided property access. Leroy Gonsalves provided helpful comments on this manuscript and Kim Colyvas provided statistical advice. Funding for this PhD project was sourced from: The Tom Farrell Institute for the Environment; Donaldson Conservation Trust; Hunter-Central Rivers Catchment Management Authority; Royal Zoological Society of NSW (Ethel Mary Read Award); Wambo Coal Pty Ltd; and Australian Geographic Society. This project was conducted under scientific licence and animal ethics approvals, under the Australian Code of Practice for the Care and Use of Animals for Scientific Purposes, 7th edition, 2004.
Appendix A: Equipment

Figure S1: PVC tower with receiver (Regal 2000) and three-element yagi antenna (Titley, Electronics, Balina, Australia) used at fixed telemetry stations and during the field trial

Figure S2: PVC tower used to support three-element yagi antenna in the field trial and during radio-tracking
Appendix B: Linear distance test

Figure S3: Station and transmitter layout for linear distance test.

The red triangle indicates the location of the receiver and PVC tower and the yellow circles indicate the different transmitter locations.

Figure S4: Average bearing error during the linear distance test

Mean bearing error (log$_{10}$ transformed) for different distances calculated during the linear distance test using three receivers and two transmitters (n = 80). Distance categories not connected by the same letter (A, B) are significantly different (α = 0.05) calculated using Tukey's HSD post-hoc tests. Error bars represent standard error.
Figure S5: Transmitter positioned on 5.1 m pole along an old pipeline route for the linear distance test
Appendix C: Location error ellipse area

![Graph showing distribution of error ellipse areas](image)

**Figure S6: Error ellipse area of triangulated bat locations**

Error ellipse area (ha) of final triangulated locations (n = 126) to four lactating female *Mormopterus norfolkensis* (M341, M342, M344, M345) during radio-tracking and that were used to create utilisation distributions for each individual.
Appendix D: Home range size asymptote graphs

a)

Average home range size

Number of locations

b)

Average home range size

Number of locations
Figure S7: Home range size asymptote graphs

Average home range size (95% volume contour, calculated from fixed kernel density estimators with a smoothing parameter of 800) for a) M341, b) M342, c) M344 and d) M345. Error bars represent standard error. For each individual, randomly selected location points were added in succession from a starting point of five locations until the maximum number of locations was reached. This process was repeated 20 times and the average calculated.
Appendix E: Stationary test bearings

Figure S8: Stationary bearing error test results

Bearings recorded from a fixed telemetry station to the fixed transmitter located in a roost tree during the radio-tracking study of Mormopterus norfolkensis. The average bearing error recorded was $8.7 \pm 0.8 \, ^\circ$ (SD $8.2 \, ^\circ$, range $0.2 - 47.2 \, ^\circ$).
### Appendix F: Resource utilisation function results

**Table S1: Resource utilisation function results**

Beta and standard errors for all habitat types in the resource utilisation function model of the utilisation distribution of four lactating female *Mormopterus norfolkensis*. Betas have been converted to absolute values of inverted percent UD volume.

<table>
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<th>CLE $\beta$</th>
<th>CLE SE</th>
<th>DSF $\beta$</th>
<th>DSF SE</th>
<th>EST $\beta$</th>
<th>EST SE</th>
<th>FWW $\beta$</th>
<th>FWW SE</th>
<th>URB $\beta$</th>
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<th>EST</th>
<th>FWW</th>
<th>URB</th>
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* Population standard error incorporates inter-animal variation, calculated using Eq 3 in (Marzluff et al. 2004).

See Figure 6-3 for habitat type abbreviations.
Chapter 7

Comparison of molecular and microscope methods of studying diet from faecal material: a case study of a threatened insectivorous bat species

Anna McConville, Leroy Gonsalves, Bradley Law and Michael Mahony
Chapter 7 Comparison of molecular and microscope methods of studying diet from faecal material: a case study of a threatened insectivorous bat species

7.1 Abstract

Despite the occurrence of bats in almost every ecosystem, diet is poorly understood for many species of insectivorous bat. Observational studies are very difficult and the traditional microscope method of dietary analysis from faecal material has been found to be biased towards hard-bodied prey items that survive mastication and digestion. Molecular methods recently applied to faecal material of insectivorous bats show great potential. However, these methods are still being developed and comparisons among techniques are particularly useful. We examined the diet of Mormopterus norfolkensis (east-coast free-tailed bat), a threatened molossid bat and used this to contrast new, developing techniques with traditional methods of prey identification using the same faecal samples. We determined the diet from faecal material separately collected from 21 free-living M. norfolkensis and found that Lepidoptera and Diptera were the most frequently recorded prey orders, with Blattodea, Coleoptera, Hemiptera and Hymenoptera also recorded. Contrary to predictions, molecular methods did not detect soft-bodied prey items (Diptera) more frequently than microscope methods. However, we did find that the results from microscope and molecular methods differed substantially in other ways. Most notably, the ranking of orders based on the frequency of occurrence may be different and this may influence conclusions made regarding the importance of different prey items. Molecular methods were able to identify prey at a much finer taxonomic resolution than we achieved using a microscope and this allowed us to explore broader ecological
concepts such as predator-prey interactions (small and tympanate prey items) and foraging habitat. However, DNA sequences were unable to be amplified for five samples and microscope methods identified prey taxa not recorded via molecular methods. Overall, our results suggest that until molecular methods can be refined further, greater sample sizes for molecular studies or a combination of methods is likely to be appropriate.

7.2 Introduction

Diet is an important component of an animal’s ecology and can provide insight into foraging behaviour, distribution, reasons for decline and possibly identify new anthropogenic threats. However, despite the occurrence of bats in almost every ecosystem, diet is poorly understood for many insectivorous bat species. They are small, fast-flying and nocturnal and this combined with the small size of their prey, makes observational studies very difficult. As a result, much of the data on insectivorous bat diet has been collected from prey remains in faecal material and stomach contents (e.g. Vestjens and Hall 1977, Kunz and Whitaker 1983, Kunz et al. 1995). The non-invasive collection of faecal material has been the favoured source of dietary information for insectivorous bats (Whitaker et al. 2009). However, the traditional method of identifying prey from faecal material using a microscope is biased towards hard-bodied prey items that survive mastication and digestion (Rabinowitz and Tuttle 1982, Kunz and Whitaker 1983, Dickman and Huang 1988) and identifications are usually restricted to broad taxonomic classifications such as order or family (e.g. Kunz et al. 1995, Rydell and Yalden 1997).

Recently, researchers have used molecular methods to identify insect prey items in the faecal material of insectivorous bats (Clare et al. 2009, Bohmann et al. 2011, Clare et al. 2011, Razgour et al. 2011a, Zeale et al. 2011, Alberdi et al. 2012, McCracken et al. 2012).
DNA from prey is extracted from faecal material and sequences compared to reference barcode libraries to identify what bats are eating. In this way, researchers are now able to address questions that have previously remained unexplored, such as the importance of small, soft-bodied prey items, such as mosquitoes (Gonsalves et al. 2013a, Gonsalves et al. 2013c). However, these methods are still being refined and comparisons among techniques are particularly useful (e.g. Zeale et al. 2011, Dodd et al. 2012).

*Mormopterus norfolkensis* Gray, 1839 is a relatively small (7 - 9 g) molossid bat, found on the east coast of Australia. It is listed as vulnerable under the New South Wales (NSW) *Threatened Species Conservation Act 1995* and as vulnerable C1 under the International Union for Conservation of Nature red list. It has a moderately low frequency and narrowband echolocation call (characteristic frequency 31 - 35 kHz; Pennay et al. 2004) and is likely to capture insect prey by aerial interception (McKenzie and Rolfe 1986, Schnitzler and Kalko 2001). The diet of *M. norfolkensis* has not been previously documented. However, other studies suggest variation among Australian molossid species (Vestjens and Hall 1977, Fullard et al. 1991, Milne 2006, Reside and Lumsden 2011).

Echolocation call design has been proposed to constrain the detection of small prey items by bats, making these small prey unavailable to bats with low frequency echolocation (Mohl 1988, Barclay and Brigham 1991). Following the calculations of Mohl (1988), the minimum prey size able to be detected by *M. norfolkensis* is 7.8 - 8.8 mm. However, medium- to large-sized bats have been found to consume small prey (Rydell 1989, Jones 1995, Waters et al. 1995) and there is much debate about the constraints imposed by echolocation on prey sizes. Swarming behaviour may make some small prey more detectable (Jones and Rydell 2003) and some studies have proposed that echolocation
call frequency is independent of detection range (Waters et al. 1995) and that call characteristics may be adjusted depending on task and habitat (Jakobsen et al. 2012).

Some insects have tympanic organs that provide ultrasonic hearing and this provides them with the opportunity to implement a defence strategy against predation by echolocating bats (Jones and Rydell 2003). The allotonic frequency hypothesis (Fullard 1987) predicts that bats calling within the hearing range of tympanate insects (20 - 50 kHz) will have fewer tympanate prey in their diet and this is supported by the diets of some bats (e.g. Rydell and Arlettaz 1994, Pavey and Burwell 1998). However, these predator-prey interactions are not fully understood and tympanate prey are consumed by some bats with echolocation calls 20 - 50 kHz (including another molossid bat species; Clare et al. 2009, Bohmann et al. 2011).

In this study, we compare a new and developing molecular method with the traditional microscope method of prey identification using faecal samples collected from *M. norfolkensis*. We predict that 1) there will be differences in the frequency of detection of soft-bodied prey items, such as Diptera (flies), between methods. We also predict that the relatively low frequency echolocation call of *M. norfolkensis* will constrain the availability of certain prey items such that: 2) small-sized (< 9 mm; Barclay and Brigham 1991) and 3) tympanate (Fullard 1987) prey items will not be consumed.

### 7.3 Methods

#### 7.3.1 Study area

The study was conducted in the Hunter Estuary, NSW, where the Hunter River enters the ocean at the Port of Newcastle (32°55’36”S 151°46’44”E; Figure 7-1). The area experiences a warm temperate climate (average monthly temperatures 8.4 – 25.6 °C) and
average annual rainfall of 1134 mm (Bureau of Meteorolgy 2012). Bats were captured for radio-tracking in the western portion of what is now a 2600 ha landmass known as Kooragang Island. Areas surrounding Kooragang Island have been modified by a long history of agriculture, coal mining and port-associated industries. The non-industrial areas of Kooragang Island are dominated by mangroves and saltmarsh, with pasture and smaller areas of freshwater wetland also occurring. Hexham Swamp, a large 2500 ha wetland dominated by freshwater reeds at the time of the study, occurs to the west. Further west of Hexham Swamp are areas of undulating hills which support young dry sclerophyll forests that overlay nutrient-poor soil.
7.3.2 Collection of bat faeces

We captured bats using harp traps (Austbat P/L, Victoria, Australia) set in mature *Avicennia marina* subsp. *australisca* (grey mangrove) forests or nearby. Faecal pellets were collected from individual *M. norfolkensis* captured for a radio-tracking study (Chapter 4). Samples analysed were from bats captured 3 - 27 January 2010, with one male (B21) captured 15 December 2009. On removal from traps, bats were held singularly in
separate clean calico bags for 1 - 14 hours until release. Faecal pellets were collected and placed into small paper envelopes, left to dry for up to 1 week and then frozen. One sample was stored in a small plastic vial and then frozen (B21). Samples with > 5 faecal pellets were analysed from 21 *M. norfolkensis* (16 females and five males).

### 7.3.3 DNA analysis

We extracted genomic DNA from a pooled sample of five faecal pellets per bat using commercial DNA kits (Ultraclean Fecal DNA kit, Mo Bio Laboratories; QIAamp DNA Stool Mini Kit; Qiagen) and amplified a region of the Cytochrome Oxidase I gene conserved in arthropods using the primers of Zeale et al. (2011) prior to cloning and following the methods of Gonsalves et al. (2013a). We used traditional polymerase chain reaction (PCR), cloning and Sanger sequencing. For each sample, sequences were entered into the identification engine on the barcoding of life database (BOLD; http://www.barcodinglife.org) and the nearest match and percent similarity of each sequence was recorded. Taxonomic assignment thresholds of Zeale *et al.* (2011) were used to assign insect DNA sequences to order, family, genus or species.

### 7.3.4 Manual identification

After DNA extraction, we examined each sample under a dissecting microscope and visually identified prey fragments from 20 samples (B21 was not analysed). We identified prey fragments to order level using a key to commonly eaten arthropods (Whitaker *et al.* 2009) as a rough guide. The relative abundance of each order in faecal samples was scored based on the number of fragments that were present: 0 (none); 1 (1 - 2 fragments); 2 (3 - 5 fragments); 3 (6 - 20 fragments); or 4 (> 20 fragments). The fragments that were most useful for identification were moth scales, antennae, hard body fragments and mouth-parts.
7.3.5 **Statistical analyses**

We tallied the total number of faecal samples that contained each prey order (frequency of occurrence) and used this to compare the microscope and molecular methods for each of the major prey orders (Coleoptera, Diptera and Lepidoptera) using a Fisher’s exact test. We also compared taxonomic richness (the number of orders recorded in each faecal sample) between microscope and molecular identification methods using a Wilcoxon signed rank test for paired data (excluding M21 which was not analysed under the microscope). To determine whether there were any sex-specific differences, we compared taxonomic richness between males and females using Wilcoxon rank sum tests. Finally, we used Wilcoxon rank sums to test for statistical difference in the relative abundance of each insect order between males and females obtained from the microscope method of prey identification. All statistical analyses were undertaken in JMP (version 9.0, SAS Institute Inc.), results are presented as mean ± standard error and significant differences are reported at $\alpha = 0.05$.

7.4 **Results**

We recorded six insect orders: Blattodea (cockroaches), Coleoptera (beetles), Diptera, Hemiptera (true bugs), Hymenoptera (wasps and ants) and Lepidoptera in the diet of 21 *M. norfolkensis*. The microscope method identified prey items from five orders and identification of fragments to finer resolution was generally unsuccessful. The molecular method identified sequences from four species, 16 genera, 31 families and four orders (Table 7-1). However, prey DNA sequences were not able to be amplified for five samples (23.8 %).
Table 7-1: Molecular prey identification results

Nearest matches of DNA sequences obtained from the faeces of *M. norfolkensis* using molecular prey identification techniques. The frequency of occurrence of each taxa in samples from male and female *M. norfolkensis* is given.

<table>
<thead>
<tr>
<th>Order</th>
<th>Family</th>
<th>Genus</th>
<th>Species</th>
<th>Common name</th>
<th>Size* (mm)</th>
<th>Male (n = 5)</th>
<th>Female (n = 16)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blattodea</td>
<td></td>
<td>cockroach</td>
<td></td>
<td></td>
<td>3 - 65</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Coleoptera</td>
<td></td>
<td>beetle</td>
<td></td>
<td></td>
<td>0.5 - 80</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Diptera</td>
<td>Chloropidae</td>
<td>grass fly</td>
<td></td>
<td></td>
<td>1.5 - 5</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Diptera</td>
<td>Culicidae</td>
<td><em>Aedes</em></td>
<td>vigilax</td>
<td>saltmarsh mosquito</td>
<td>4</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Diptera</td>
<td>Culicidae</td>
<td><em>Aedes</em></td>
<td></td>
<td>mosquito</td>
<td>3 - 6</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>Diptera</td>
<td>Pipunculidae</td>
<td><em>Cephalops</em></td>
<td></td>
<td>big-headed fly</td>
<td>3 - 6</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Diptera</td>
<td>Psychodidae</td>
<td><em>Sergentomyia</em></td>
<td></td>
<td>sand fly</td>
<td>3 - 5</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Diptera</td>
<td>Tachinidae</td>
<td><em>Hyphantrpha</em></td>
<td></td>
<td>tachinid fly</td>
<td>5 - 35</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Diptera</td>
<td>Tipulidae</td>
<td>crane fly</td>
<td></td>
<td></td>
<td>6 - 75</td>
<td>1</td>
<td>3</td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Arctiidae</td>
<td><em>Termessa</em></td>
<td></td>
<td>lichen moth</td>
<td>7 - 90</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Coleorphoridae</td>
<td><em>Coleophora</em></td>
<td>alcyonipennella**</td>
<td>clover casebearer</td>
<td>6 - 21</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Geometridae</td>
<td><em>Dysbatus</em></td>
<td>singularis</td>
<td>looper</td>
<td>7 - 90</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Geometridae</td>
<td><em>Dysbatus</em></td>
<td></td>
<td>looper</td>
<td>7 - 90</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Noctuidae</td>
<td><em>Ericcia</em></td>
<td></td>
<td>moth</td>
<td>7 - 90</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Noctuidae</td>
<td><em>Mythimna</em></td>
<td>convecta</td>
<td>common armyworm</td>
<td>35 - 40</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Oecophoridae</td>
<td><em>Antipterna</em></td>
<td></td>
<td>moth</td>
<td>11 - 32</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Oecophoridae</td>
<td><em>Oligoloba</em></td>
<td></td>
<td>moth</td>
<td>20 - 28</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Oecophoridae</td>
<td><em>Philobota</em></td>
<td></td>
<td>tunnel moth</td>
<td>15 - 38</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Tortricidae</td>
<td><em>Homonia</em></td>
<td></td>
<td>leaf roller</td>
<td>7 - 40</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>Xyloryctidae</td>
<td><em>Cryptophasa</em></td>
<td></td>
<td>timber moth</td>
<td>7 - 90</td>
<td>2</td>
<td></td>
</tr>
</tbody>
</table>
Overall, Lepidoptera and Diptera were the most frequently recorded orders in the diet of *M. norfolkensis* (Figure 7-2). There were differences in the ranking of the frequency of occurrence of prey orders between analysis methods, with the microscope method detecting Lepidoptera most frequently, whereas the molecular method recorded Diptera marginally more frequently than Lepidoptera (Figure 7-2). Lepidoptera occurred in the diet of 85% of individuals using the microscope method and 56.5% of individuals using the molecular method (Figure 7-2). Lepidoptera recorded comprised 10 different genera, with an estimated size range of 6 - 90 mm (Table 7-1). It was not possible to be more specific about the size of most Lepidoptera recorded as there was considerable size variation within each genus (often the finest resolution of identification obtained; Table 7-1). Two tympanate families of Lepidoptera were consumed, Geometroidea and Noctuoidea (Table 7-1) and these were recorded in the diet of five individuals (23.8%), which was similar to the occurrence of non-tympanate moth families (six individuals; 28.6%).
Figure 7-2: Frequency of occurrence of prey orders

Frequency of occurrence of insect prey in *Mormopterus norfolkensis* diet obtained using a) microscope and b) molecular methods of prey identification from faecal material. Sample sizes are different as one sample (M21) was not analysed under the microscope. LEP - Lepidoptera; DIP - Diptera; COL - Coleoptera; HEM - Hemiptera; HYM - Hymenoptera; BLA - Blattodea.
Diptera was also frequently recorded in the diet of *M. norfolkensis*, occurring in 62.5 % of samples when analysed using the molecular method and 35 % of samples analysed using the microscope method (Figure 7-2). Whilst only one Diptera family (Chironomidae, non-biting midges; 2 - 4 mm) was recorded via microscope analysis, six families were recorded using molecular methods, including a number of very small taxa (1.5 - 6 mm) such as *Aedes* (mosquitoes), *Cephalops* (big-headed flies), *Sergentomyia* (moth flies) and *Chloropidae* (grass flies; Table 7-1).

Coleoptera were recorded in the diet of a total of six individuals, two via the molecular method and a different four individuals using the microscope method (Figure 7-2). Blattodea was recorded in the diet of two individuals using the molecular method and was not detected using the microscope method (Figure 7-2). Hymenoptera and Hemiptera appeared to be the least important orders, recorded in the diet of only one individual each and only using the microscope method (Figure 7-2).

Whilst there was no significant difference in the frequency of occurrence of Coleoptera and Diptera between microscope and molecular methods (*p > 0.4099*), microscope methods recorded Lepidoptera more frequently than molecular methods (*p = 0.0088*). However, overall there was no significant difference in taxonomic richness between the microscope (1.55 ± 0.17) and molecular methods (1.1 ± 0.19; *W = -29, p = 0.1155; n = 20*). Whilst there was a trend for samples from females to have a slightly greater taxonomic richness using the microscope method (1.69 ± 0.20, *n = 16*) than recorded for males (one taxa recorded per individual; *n = 4*; *W = 26, p = 0.0944*), there was no significant difference between taxonomic richness of male (1.20 ± 0.37; *n = 5*) and female (1.06 ± 0.21; *n = 16*) diets using the molecular method (*W = 60; p = 0.6898; Figure 7-3*). There was also no significant difference in the relative abundance of different prey orders in individual samples between males (*n = 5*) and females (*n = 15*) obtained using a
microscope (Diptera: $W = 39.5$, $p = 0.8300$; Coleoptera: $W = 34$, $p = 0.3092$; Lepidoptera: $W = 44$, $p = 0.8676$; Figure 7-4). See Appendix 1 for a comparison of prey orders recorded in each sample.

Figure 7-3: Prey taxonomic richness in individual samples

Taxonomic richness (number of prey orders) of prey recorded in Mormopterus norfolkensis male and female faecal material using a) microscope and b) molecular methods of prey identification.
Figure 7-4: Relative abundance of prey orders in individual samples via microscope

Relative abundance of Diptera, Coleoptera and Lepidoptera recorded via microscope analysis of male (n = 5) and female (n = 15) Mormopterus norfolkensis faecal material. Relative abundance of each major prey taxa were scored using an ordinal scale based on the number of prey items in faecal material, where 0 = none, 1 = 1 - 2 fragments, 2 = 3 - 5 fragments, 3 = 5 - 20 fragments and 4 = > 20 fragments. Due to differences in detectability and fragment size, direct comparisons among prey orders are not appropriate.
7.5 Discussion

This is the first study to investigate the diet of *M. norfolkensis* and we found that Lepidoptera and Diptera were the most frequently encountered prey taxa, with Coleoptera also regularly occurring. Contrary to prediction 1, there was little difference in the frequency of detection of soft-bodied prey items (Diptera) between methods. However, we did find that the results from microscope and molecular methods differed substantially in other ways. Most notably, the ranking of orders based on the frequency of occurrence were different. Molecular methods were able to identify prey at a much finer taxonomic resolution than we achieved using a microscope and this allowed broader ecological concepts to be explored such as predator-prey interactions (small and tympanate prey items) and foraging habitat. However, DNA sequences were unable to be amplified for some samples and microscope methods identified prey taxa not recorded via molecular methods. We found no evidence to suggest that *M. norfolkensis* partitions foraging resources between sexes during the maternity season. However, we had relatively few male samples (five) and it is possible that subtle differences do occur between the sexes that were undetected in our study.

7.5.1 Comparison of methods

Our finding that the ranking of prey orders based on the frequency of occurrence may depend on the method used to analyse faecal material is important for future studies to consider. For example, if we were to rely on the microscope analysis alone, we would have concluded that *M. norfolkensis* foraged primarily on Lepidoptera, whereas the molecular methods indicate that Diptera and Lepidoptera were equally important dietary components. Whilst we found that there was no significant difference in the frequencies of detection of soft-bodied prey (Diptera) between methods, the two methods rarely detected Diptera in the same samples, suggesting deficiencies in both methods. For example,
Despite recording the distinctive antennae of male Chironomidae within the faecal material of eight individuals under the microscope (one male bat with over 20 antennae fragments), no DNA sequences from this family were obtained from any sample using molecular methods. While we found that Lepidoptera was recorded more frequently using microscope than by molecular methods, this is not unexpected as Lepidoptera scales are easily identified via microscope and can remain in the digestive system of bats for a long period (32 hours after consumption or in 59 subsequent droppings; Robinson and Stebbings 1993).

One of the major limitations of the molecular method in our study is the relatively few prey taxa identified per sample. The molecular methods applied in this study used traditional PCR, cloning and Sanger sequencing. It is possible that prey with relatively low concentrations of DNA within faecal samples were not detected due to PCR bias in which DNA at low starting concentrations may become exponentially less likely to be amplified in subsequent PCR cycles. Furthermore, a greater sample size of sequences for each clone library (> 16 clones) may have provided a greater number of identified prey taxa. More recent molecular dietary studies have employed next-generation sequencing that can overcome this limitation encountered (Bohmann et al. 2011, Razgour et al. 2011a) and these techniques are becoming increasingly more affordable. It should also be noted that other studies have found that the DNA barcoding library used to identify DNA sequences may also influence results (Dodd et al. 2012). Further investigation into how reliably DNA can be extracted from different prey fragments (including moth scales) would be valuable for future studies.

Another major limitation with current molecular methods is that the relative amount of each prey item consumed remains difficult to quantify. Some studies have indicated that
clone sequence proportions may be used to infer the relative amount of each prey item consumed (Sutherland 2000, Zeale et al. 2011). However, variability has been found in both the clone sequence proportions (L. Gonsalves, unpubl. data) and DNA degradation rates (Deagle and Tollit 2007), indicating that the relationship between clone sequence and the amount of prey consumed is unreliable. Yet, being able to quantify the proportion of prey items consumed is of great importance to prey selection studies and further development of methods to address this issue is required.

7.5.2 Fine resolution of molecular methods to infer foraging habitat

Prey items were able to be identified at a fine taxonomic resolution using molecular methods and this provides more information than traditional microscope methods (Razgour et al. 2011a). In other studies that have species- or genus-level identifications, conclusions have been made about the type of foraging habitat used (Clare et al. 2011, Alberdi et al. 2012). While some of the prey items recorded in the diet of *M. norfolkensis* in our study occur in a wide range of habitats, such as the Chloropidae (grass flies) and *Homona* sp. (leaf rollers), other prey taxa have more restricted habitat requirements and this may provide some insight into the foraging habitats used by *M. norfolkensis*. *Coleophora alcyonipennella* (clover casebearer) is an introduced moth species that feeds on the nectar of *Trifolium* sp. (clover; a common herbaceous plant in pastures) and the native moth *Mythimna convecta* (common armyworm) is an agricultural pest species with larvae that feed on pasture and lawns (Common 1990). The presence of these two prey items may suggest that *M. norfolkensis* was foraging near pasture at some stage prior to capture (providing adults don't disperse over other areas). These cleared areas were found to occur within the home ranges of radio-tracked *M. norfolkensis* from the same capture site (Chapter 6). However, cleared areas were used significantly less than other vegetation types, particularly above freshwater wetland in the radio-tracking study.
(Chapter 6) and the dietary results presented here may place more emphasis on these cleared areas than is appropriate.

The complex life-cycle requirements of insects also makes interpretation difficult, as adult prey may have been captured during dispersal and *M. norfolkensis* foraging habitats may not necessarily coincide with the preferred habitats of the prey item. For example, some of the other prey taxa recorded have aquatic larvae (*Sergentomyia* sp., Tipulidae and Chironomidae) and adults may have been captured as they emerged or during mating swarms in moist areas, such as from the freshwater wetland. However, they could have also been captured during dispersal. We also recorded two parasitic taxa (*Hyphantrophaga* sp. and *Cephalops* sp.) which may have been captured by *M. norfolkensis* as adults or their larval stages may have been parasitising prey items captured. The ecology of many invertebrates is poorly understood and whilst there are some consistencies between diet and the habitat use findings from home range studies of lactating female *M. norfolkensis*, these remain speculative. Therefore, interpretation of foraging habitat based on the ecology of prey taxa alone is speculative and any conclusions regarding bat foraging habitat should be made with caution and ideally, verified with other techniques, such as radio-tracking.

### 7.5.3 Small prey items

We found small Diptera in the diet of *M. norfolkensis* and this contrasted with prediction 2 and the notion that small prey are unavailable to bats which have low frequency echolocation (Barclay and Brigham 1991). However, the small Diptera (such as *Aedes* sp. and Chironomidae) found in the diet of *M. norfolkensis* may have been swarming or captured in locations where they were superabundant, making these tiny prey items easier to detect (given that the echo target strength of a swarm is higher than for
individual prey; Waters et al. 1995) and energetically efficient to capture. However, at least two other small Diptera taxa were consumed by *M. norfolkensis*, with one sample containing *Sergentomyia* sp. (sand flies thought to feed on reptiles) and another containing Chloropidae. The presence of these non-swarming and non-parasitic Diptera suggests that there is a possibility that *M. norfolkensis* regularly captures small prey items. Recent studies have suggested that bats may adjust the characteristics of their echolocation calls to provide an optimal field of view (Jakobsen et al. 2012). In this way, small prey items may still be available to low-frequency echolocating bats, providing that they focus their calls accordingly. Small Diptera have also been found in the diet of other molossid bats with low frequency echolocation calls (Bohmann et al. 2011), providing some support to this idea. We were unable to identify any Diptera other than Chironomidae via microscope methods and this inability to accurately quantify prey abundance limits our ability to draw firm conclusions regarding the importance of Diptera to *M. norfolkensis*.

7.5.4 Tympanate prey

The molecular methods provided fine resolution information regarding Lepidoptera, which are difficult to identify beyond order-level using microscope methods (Whitaker et al. 2009). Contrary to prediction 3, we found that the frequency of occurrence of tympanate moths in the diet of *M. norfolkensis* individuals was similar to the occurrence of non-tympanate moths, despite *M. norfolkensis* having an echolocation call within the hearing range of most tympanate insects. Tympanate prey items have also been found in the diet of other low frequency echolocating molossid bats (Bohmann et al. 2011) and it is possible that these tympanate moth species were particularly abundant or that bats were hunting in groups (Dechmann et al. 2010), making capture easier.
We found that *M. norfolkensis* consumed a range of prey taxa and this tentatively suggests that this threatened species is unlikely to be a dietary specialist. However, our study was limited in spatial and temporal scope and we also found relatively few taxa per sample. As such, it is likely that we only sampled a small portion of the insect diversity consumed by *M. norfolkensis*. However, future studies may address the issue of low identification rates of prey taxa when using molecular methods through the development of more robust DNA extraction methods and increased sample sizes for molecular studies. Research into ways to quantify the amount of each prey item consumed, when using molecular methods, would be of particular value. Until this occurs, a combination of molecular and microscope methods is likely to be appropriate for dietary studies (Zeale et al. 2011, Dodd et al. 2012).

### 7.6 Acknowledgements

We would like to thank: Lisa Schroeter and Margarete Überfuhr for assisting with the microscope lab work; Dave Britton for assisting with some prey fragment identifications; and the volunteers that helped with field work. Funding for this PhD project was awarded to AM by: The Tom Farrell Institute for the Environment; Donaldson Conservation Trust; Hunter-Central Rivers Catchment Management Authority; Royal Zoological Society of NSW (Ethel Mary Read Award); Wambo Coal Pty Ltd; and Australian Geographic Society. This project was conducted under scientific licence and animal ethics approvals, under the *Australian Code of Practice for the Care and Use of Animals for Scientific Purposes, 7th* edition, 2004.
Appendix A - Prey order identification per sample

Table S1: Presence (√) of insect orders in the diet of *Mormopterus norfolkensis* as determined by analysis of faecal material using molecular barcoding (DNA) and microscope (MIC) methods.

<table>
<thead>
<tr>
<th>Bat ID</th>
<th>Date</th>
<th>Sex</th>
<th>Blattodea DNA</th>
<th>Blattodea MIC</th>
<th>Hymenoptera DNA</th>
<th>Hymenoptera MIC</th>
<th>Lepidoptera DNA</th>
<th>Lepidoptera MIC</th>
<th>Diptera DNA</th>
<th>Diptera MIC</th>
<th>Hemiptera DNA</th>
<th>Hemiptera MIC</th>
<th>Coleoptera DNA</th>
<th>Coleoptera MIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>B1⁺</td>
<td>12/01/2010</td>
<td>F</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>√</td>
<td>-</td>
<td>-</td>
<td>-</td>
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<td>3/01/2010</td>
<td>F</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>√</td>
<td>√</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>√</td>
</tr>
<tr>
<td>B3</td>
<td>3/01/2010</td>
<td>F</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>√</td>
<td>√</td>
<td>-</td>
<td>√</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
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<td>-</td>
<td>-</td>
<td>-</td>
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<td>F</td>
<td>√</td>
<td>-</td>
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<td>-</td>
<td>-</td>
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<td>√</td>
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⁺ Individuals where no prey DNA sequences were able to be amplified from samples

⁻ Samples not analysed under the microscope
Chapter 8

General Discussion
Chapter 8 General Discussion

In this thesis, key aspects of the ecology of *M. norfolkensis* were investigated to provide evidence-based conservation management strategies and impact assessments. I have rigorously explored the habitat of *M. norfolkensis* within the Hunter Region at multiple spatial scales, discovered important maternity roosts in mangrove forest and undertook the first dietary study of *M. norfolkensis*. While these studies were focused on *M. norfolkensis*, they also contribute to global ecological concepts. For example, I contrast three morphologically similar species and determine whether microhabitat use predictions can be extrapolated to trends in landscape-scale habitat use. I provide a comparison of novel molecular and traditional microscope methods of prey identification in faecal material, which provides guidance for future use of molecular methods. I also provide a case study in the validation of habitat models at different spatial scales.

8.1 Synthesis of Key Findings

Habitat use was investigated at multiple spatial scales, using three independent and systematically collected datasets (Chapters 2, 5 and 6) and overall, productive floodplain areas are likely to be preferred habitat for *M. norfolkensis* in the Hunter region, with urban land-use and dry sclerophyll forest avoided. This contrasts with the habitat description of dry sclerophyll forest and woodland given by field guides (Churchill 2008, Hoye et al. 2008) and illustrates that systematic studies are required to fully explore habitat use as anecdotal evidence of highly mobile species may be generalised or mis-leading. Additionally, Chapter 2 illustrates that habitat selection is likely to be a result of many different competing factors and it can be difficult to make detailed predictions of broad-
scale habitat use based on coarse-grained morphological adaptations and echolocation call design characteristics (Luck et al. 2013).

Few studies test the performance of habitat models against systematically collected and independent datasets (Manel et al. 2001, Araujo and Guisan 2006, Mouton et al. 2010, Razgour et al. 2011b). Yet, it is important to quantify the predictive ability of habitat models to ensure their appropriate use (Elith et al. 2002). In Chapter 5, the validity of regional-scale habitat models (Chapter 2) was assessed at a local-scale and the errors visualised. Broad habitat types that were predicted from local-scale models (activity level data, Chapter 5) were generally consistent with those from regional-scale models (presence / absence data, Chapter 2). However, threshold-dependent accuracy measures indicated a poor fit, with both false positives and false negatives occurring. As such, caution should be applied when using the regional models at a fine scale, particularly when the consequences of fine-scale errors to conservation are severe.

At the finest-scale, resource utilisation functions (RUFs) were used to investigate habitat use within the home range of lactating female *M. norfolkensis* (Chapter 6). Again, the habitat use findings were broadly consistent with predictions from regional and local-scale habitat models. However, despite predictions of highly suitable habitat in estuarine areas by regional- and local-scale models, the use of estuarine habitats was inconsistent among radio-tracked individuals. This inconsistency in use may be explained by the presence of roosting habitat within the mangrove forests, with foraging activity focused over freshwater wetland.

The rarity of *M. norfolkensis* was explored at a broad-level by contrasting *M. norfolkensis* habitat use with two other morphologically similar and sympatric species (*Mormopterus* species 2 and *Mormopterus* species 4) and found that, despite similarities in morphology
and echolocation call design, the three species used different habitats (Chapter 2). *Mormopterus* species 4 had a vastly different distribution (with little indication of a preference for particular habitat elements and a tolerance for low density urban centres), whereas *M. norfolkensis* and *M. species 2* used more similar coastal environments. Most importantly, despite their overlap in use of productive floodplain areas, *M. norfolkensis* avoided urban areas, whereas *M. species 2* was frequently recorded in urban habitat types at both regional- (Chapter 2) and local-scales (Chapter 5). These findings suggest that, in the Hunter Region, *M. norfolkensis* is sensitive and *M. species 2* is more tolerant of urbanisation, consistent with a study in Sydney (Threlfall et al. 2012). The extensive use of coastal floodplains by the threatened *M. norfolkensis* (Chapters 2, 5 and 6) is significant because these habitats are under increasing pressure from human land-uses with the predicted increase in urbanisation is likely to further reduce the amount of suitable habitat. The Lower Hunter Regional Strategy in 2006 estimated that 115,000 new dwellings (including up to 69,000 new greenfield lots) will be required to support 160,000 additional people in the region by 2031 (NSW Department of Planning 2006).

It can be difficult to obtain rigorous scientific data at the individual-level for highly mobile and rarely-captured species and Chapter 3 illustrates this point, with only six individuals being able to be captured for radio-tracking despite substantial effort. However, regardless of limited sample size, this study recorded unexpected roosting behaviour by *M. norfolkensis* in a rainforest gully and telegraph pole and provided observations of foraging in productive alluvial flats in two different agricultural areas.

A large population of *M. norfolkensis* was subsequently discovered roosting in mangrove forest that was conducive to trapping and this facilitated rigorous data collection regarding roost selection (Chapter 4) and home range (Chapter 6). Lactating females were faithful to two patches of mangrove forest close to where they were captured, regularly switched
roosts and roosted in hollows singularly or in small groups. Lactating female *M. norfolkensis* had very large (4,099.5 ± 528.6 ha) and overlapping home ranges. Maternity roosts were located in locally unique mangrove forests that had abundant hollow-bearing trees (an astounding 837.0 ± 47.3 hollow-bearing stems per hectare) and a stable microclimate compared with other available habitats. It was suggested that lactating females may benefit from the operation of a fission-fusion society among the maternity colony as a whole.

Finally, in Chapter 7, the diet of *M. norfolkensis* was investigated and the traditional microscope method of prey identification from faecal material was compared with a new and developing molecular method. Overall, Lepidoptera and Diptera were the most frequently encountered orders in the diet of 21 *M. norfolkensis*, with Blattodea, Coleoptera, Hemiptera and Hymenoptera also recorded. *Mormopterus norfolkensis* consumed a range of prey taxa and this tentatively suggests that this rare species is unlikely to be a dietary specialist. However, it is likely that only a small portion of the diversity consumed by *M. norfolkensis* was sampled, as relatively few taxa were found per faecal sample.

### 8.2 Conservation status review

There is an emerging shift in focus from species-level to ecosystem-level conservation and research priorities in Australia (Lunney et al. 2011a). However, *M. norfolkensis* is one example of a species that required targeted study to develop a detailed understanding of its ecology. Previous systematic studies of bat communities and functional relationships failed to model habitat use by *M. norfolkensis* due to lack of records (Basham et al. 2010, Threlfall et al. 2011, Luck et al. 2013). Other studies focused on particular landscape elements and were able to obtain only a partial representation of *M. norfolkensis* habitat.
use (e.g. Laegsgaard et al. 2004, Lloyd et al. 2006). The targeted studies presented in this thesis illustrate that species-level research is required, particularly for rare species, as in addition to low samples sizes, unexpected behaviours are unlikely to be revealed in a study with broader objectives. For example, without the substantial time spent searching for sites to reliably capture *M. norfolkensis*, the use of mangroves as maternity roosts would not be known. Without multi-scale systematic studies to investigate and validate habitat use by *M. norfolkensis*, habitat in the Hunter Region would still be described as dry sclerophyll forest and woodland. The use of productive floodplain areas requires vastly different conservation management strategies than for forests on poorer soils. For example, less than 30 % of floodplain vegetation communities remain in NSW, whereas the forest reserves of eastern Australia contain large tracts of ‘dry sclerophyll forest and woodland’ (Keith 2004). Furthermore, based on the morphology and echolocation call characteristics of *M. norfolkensis*, without targeted study, we would continue to expect that urban areas represent habitat and this has serious implications for environmental impact studies of urban areas encroaching on bushland.

In this thesis, I have presented new data on the ecology of *M. norfolkensis* and further discussion of conservation status is appropriate. In Appendix 1, I assessed the conservation status of *M. norfolkensis* against the IUCN red list criteria and unexpectedly found that it did not meet any of the criteria for listing as a threatened species. Of most relevance, the IUCN red list criteria specified a minimum loss of habitat of ≥ 30 % over a time period of 10 years or three generations for listing as threatened (IUCN Standards and Petitions Subcommittee 2013). Within the area predicted to be suitable habitat for *M. norfolkensis* based on a MaxEnt (maximum entropy) model (Appendix 1), urban land-use increased by 5 % between 2006 and 2011. However, when viewed as a loss of *M. norfolkensis* habitat over 10 years, this equated to < 1 % reduction in the total area of
suitable habitat mapped, far less than required for listing as threatened under the IUCN red list (Appendix 1).

The general paucity of historical data on distribution or population size for Australian bats makes quantifying historical decline difficult (Lunney et al. 2011a) and this is certainly the case with *M. norfolkensis*. The productive floodplain areas that are likely to be preferred by the species have been modified by historical clearing for agriculture. However, in Chapter 2, I found that *M. norfolkensis* was more likely to occur in landscapes with less forested vegetation cover indicating it is somewhat tolerant of land-clearing. Yet, I also found that four radio-tracked individuals (Chapter 6) flew above freshwater wetland significantly more than cleared agricultural areas. Therefore, it is possible that agricultural areas may actually be marginal habitat, with remnant vegetation types (in areas of high productivity) preferred when available. Additionally, it is possible that *M. norfolkensis* occurs at low density across the landscape, either naturally or in response to habitat modification and some records represent commuting bats with large home ranges. As the detection probability of *M. norfolkensis* is unknown, the area of occupancy, calculated in Appendix 1, may not accurately represent *M. norfolkensis* population size. It is also likely that *M. norfolkensis* density is not even across all occupied areas and this may change seasonally (e.g. the formation of maternity colonies).

The widespread loss of hollow-bearing trees in both agricultural and forested areas (Gibbons and Lindenmayer 2002) may also place additional pressures on local populations through interspecific competition and the quality of cavities available for critical maternity roosts may influence reproductive success. Furthermore, the protection of *M. norfolkensis* habitat is uncertain, with < 5 % of suitable habitat (> 0.5 probability of occurrence; Chapter 2) located within conservation reserves. This emphasises that the long-term protection of *M. norfolkensis* habitat is uncertain. It is also possible that
impending impacts from climate change may place the species at greater risk. Sea level rise may reduce the amount of habitat available to *M. norfolkensis* by inundating coastal floodplain areas such as mangrove forest and freshwater wetland. These factors could have serious implications for the species, but are difficult to assess within the context of the IUCN guidelines.

Thus, it is clear that despite substantial advances in our understanding of key aspects of the ecology of *M. norfolkensis*, we still require information regarding trends in population size or habitat use over time to assess conservation status more rigorously. This is unlikely to be obtained from short-term studies, but requires a long term (at least 10 years) monitoring commitment. Yet, as illustrated by the recent extinction of *Pipistrellus murrayi* (Christmas Island pipistrelle), monitoring alone will not improve conservation status. Decisive action must accompany monitoring, in an adaptive management framework based on rigorous scientific data. There are indications that conservation concern for *M. norfolkensis* is warranted as it avoids urban areas and the coastal floodplains it occupies have been subject to broad-scale land-clearing and removal of hollow-bearing trees. In addition, productive floodplains continue to be subject to anthropogenic pressures. Therefore, a listing of ‘Near Threatened’ for *M. norfolkensis* is likely to be appropriate until further monitoring can be undertaken. The category of Near Threatened is defined as a taxon that has been evaluated against the criteria but does not qualify as threatened now, but is close to qualifying for or is likely to qualify for a threatened category in the near future (IUCN Standards and Petitions Subcommittee 2013).

### 8.3 Management implications and further research

Many of the findings of studies within this thesis have direct relevance to conservation both for *M. norfolkensis* in Australia and also broader implications for insectivorous bat
conservation globally. The key contributions of this thesis to conservation are outlined below with specific recommendations, as appropriate.

8.3.1 Specific Mormopterus norfolkensis conservation management strategies:

Applicability of our findings to other regions
Together, the studies contained within this thesis indicate that, a mosaic of different habitat types is required by *M. norfolkensis* within the Hunter Region, particularly productive floodplain habitats, along with nearby patches of forest with abundant hollows. Preliminary observations (Chapter 3) and a study in northern NSW (Law et al. 2000) indicate this is likely to be the case elsewhere in NSW. However, validation of my habitat use predictions is required in other regions (Whittingham et al. 2005, Rhodes et al. 2007, McAlpine et al. 2008, Murray et al. 2011). The roost characteristics of maternity colonies in other regions should also be investigated to assist with more accurate prediction of roost occurrence and particularly, to inform impact assessments.

Hollow-bearing tree retention
The following recommendations regarding the protection of roosts are expected to benefit *M. norfolkensis*:

- Hollow-bearing trees within 5 km of floodplains and particularly, areas with abundant hollow-bearing trees (when compared to the surrounding 5 km) should be given high conservation priority.
- Targeted surveys to identify *M. norfolkensis* maternity roosts within or in close proximity (5 km) to productive floodplain areas should be undertaken for development proposals to ensure that impacts on these critical habitat features are adequately considered. As a minimum, observation of potential roosts with an
ultrasonic detector (for species identification) should be undertaken on dusk during the maternity season in these areas.

**Long-term monitoring**

Whilst this project has substantially advanced our understanding of *M. norfolkensis* ecology, data that allows a robust assessment of conservation status is still required. In particular, long-term monitoring of changes in either the area of habitat or population size is appropriate (such as could be obtained using occupancy modelling; MacKenzie et al. 2006). In this way, other Australian insectivorous bat species that also require rigorous assessment (Lunney et al. 2011a) may be monitored simultaneously. Additionally, predictive maps have been found to be useful in targeting survey effort (Razgour et al. 2011b) and the regional-scale habitat maps created in Chapter 2 may be used to identify monitoring sites in the Hunter and Central Coast regions (see regional predictive map recommendations below).

**Roost and patch fidelity**

Some fission-fusion bat societies have been found to be stable (Kerth et al. 2011) and maternity colonies found to be loyal to roosting areas over the long-term (Willis et al. 2003, Popa-Lisseanu et al. 2008). Studies that investigate roost site (or patch) fidelity over the long term would be particularly valuable to determine the consequences of roost removal to *M. norfolkensis* (and in general for other Australian bat species).

**Importance of mangrove forests**

The Hunter Estuary mature mangrove forest is unique in the local area, having an exceptionally high abundance of hollow-bearing trees. *Mormopterus norfolkensis* maternity colonies have been recorded using a range of forest types (see Chapter 3) and *M. norfolkensis* is unlikely to be a mangrove-obligate. However, investigations into the use
of mangrove forests by *M. norfolkensis* in other parts of its range and also other areas of the Hunter Estuary would clarify the broader importance of mangrove forests to *M. norfolkensis*. A suitable study site may be in the mangrove forests of Port Stephens, which occurs to the north of the Hunter Estuary, as it has the largest area of mangrove forest in NSW (202 ha; Geoscience Australia 2012). Additionally, the importance of the Hunter Estuary mangrove forest to other insectivorous bat species should also be investigated, as captures (Chapter 4) suggested a high diversity of hollow-roosting species.

Finally, studies that quantify predation risk and hollow competition (e.g. Threlfall et al. 2013b) in mangrove forests may provide additional insight into why mangrove forests were so important for *M. norfolkensis* maternity colonies in the Hunter Estuary and would provide valuable insights about the influence of predation on roost selection insectivorous bats in general.

**Wetland restoration**

At a local-scale, the restoration of drained wetlands (through the removal of drains and encouragement of native wetland vegetation) has been found to have a positive effect on bat communities elsewhere (Menzel et al. 2005). Whilst the cleared agricultural land surrounding Hexham Swamp was used at a moderate level by *M. norfolkensis* (Chapters 5 and 6), significantly greater use was recorded over the freshwater wetland vegetation community. Therefore, the restoration of key highly productive landscape elements, such as wetlands that may have been historically converted to pasture, may increase preferred habitat for *M. norfolkensis*. This may be particularly important where roosting habitat (forests with abundant hollows) occurs nearby. To facilitate this, the regional habitat mapping (Chapter 2) may be used to identify suitable restoration areas within the Hunter and Central Coast regions (see regional predictive map recommendations below).
Insectivorous bat monitoring should be incorporated into projects that alter hydrological conditions (e.g. Hexham Swamp rehabilitation) as changes in the abundance and assemblage of aerial invertebrates is likely to affect insectivorous bat communities. By simultaneously monitoring insectivorous bats and aerial invertebrates over time, potential impacts on bat communities may be quantified and these studies would provide guidance for future rehabilitation projects elsewhere or be incorporated into an adaptive management framework.

**Urban areas**

Habitat modelling at three different spatial scales indicated that *M. norfolkensis* avoids urban areas (Chapters 2, 5 and 6) in the Hunter Region and this is consistent with findings from other studies (Basham et al. 2010, Threlfall et al. 2012, Luck et al. 2013). These findings indicate that urbanisation of productive areas should be specifically avoided to minimise further habitat loss for *M. norfolkensis*. Studies that investigate why *M. norfolkensis* is sensitive to urbanisation are also required. Of particular value, is research that investigates prey availability and disturbance (noise and lighting), in productive floodplain areas over a gradient of urbanisation.

**Climate change**

Climate change has the potential to impact *M. norfolkensis* through the loss of mangrove forest and inundation of coastal freshwater wetlands. Strategies to address these potential impacts must be prepared now and implemented at a broad planning level. For example, specific provisions need to be made to allow mangroves to colonise new locations gradually over time, so that new mangroves are old enough to form hollows by the time that current forests are lost. Recent climate change predictive modelling studies of the
Hunter Estuary wetlands will assist the formation of such strategies (Rogers et al. 2013a, b).

**Use of regional habitat maps**

While the regional-scale habitat maps (Chapter 2) were found to be inappropriate for fine-scale planning where the consequence of errors is high (e.g. for the development or conservation of individual land parcels), these maps are still a useful tool for conservation planning within the Hunter region. For example:

- A review of the regional *M. norfolkensis* habitat map found that only 4.5% of the suitable habitat (> 0.5 probability of occurrence) predicted for *M. norfolkensis* occurred within conservation reserves (as of 2012). This suggests that *M. norfolkensis* conservation strategies need to be targeted towards private landholdings, rather than existing national parks estate.

- The habitat map may be used to identify potential wetland areas that may benefit from restoration activities (see wetland restoration recommendations above) or to locate suitable long-term monitoring sites (see long-term monitoring recommendations above).

**8.3.2 Global insectivorous bat conservation:**

Chapter 4 is one of few international bat-mangrove studies (see Luther and Greenberg 2009 for a review), with published studies restricted to the extensive mangrove forests of Brazil (Bordignon 2006, Andrade et al. 2008) and north-western Australia (McKenzie and Rolfe 1986, McKenzie and Bullen 2012) and it is possible that the importance of mangrove forests to insectivorous bats may have also been overlooked in other regions of the world. Further research into the value of mangrove forests to terrestrial fauna is required globally.
Riparian areas such as rivers, waterbodies and wetlands have been found to be important to a range of insectivorous bat species (Racey and Swift 1985, Grindal et al. 1999, Holloway and Barclay 2000, Monadjem and Reside 2008, Popa-Lisseanu et al. 2009). The *M. norfolkensis* habitat use studies presented here provide further support for the conservation of productive riparian areas globally.

Chapter 5 illustrated that broad habitat type (based on vegetation community) classifications can be important predictors. I suggest that they are more practical for conservation (providing that they are good predictors) than complex combinations of raw variables as they are easily communicated to land managers.

Chapter 7 illustrated that future dietary studies need to address the issue of low identification rates of prey taxa using molecular methods through the development of more robust DNA extraction methods, increased sample sizes for molecular studies or by using a combination of molecular and microscope methods. More recent molecular dietary studies (Bohmann et al. 2011, Razgour et al. 2011a) have employed next-generation sequencing that while more expensive, may overcome the biases associated with traditional sequencing methods. Further investigation into how reliably DNA can be extracted from different prey fragments (including moth scales) would be valuable to ensure appropriate conclusions are made. Finally, the development of molecular methods that can quantify the relative proportions of each prey item consumed would be a substantial advancement for use in prey selection studies.

### 8.4 Conclusion

This thesis was initiated with the aim to broaden our understanding of the ecology of *M. norfolkensis* to better inform conservation management strategies and impact
assessment. I undertook studies that were designed not only to bridge critical knowledge gaps regarding *M. norfolkensis* ecology, but also to address broader issues in the fields of landscape and functional ecology. I explored how morphologically similar bat species use habitat at a landscape scale when they occur in sympatry. I provided a case study comparing different methods of dietary analysis which will assist future dietary studies. I also demonstrated how to apply resource utilisation functions to bats for the first time, enabling habitat use within individual home ranges to be more rigorously investigated. Habitat for *M. norfolkensis* in the Hunter Region is now well-defined and I have demonstrated that the mangrove forests of the Hunter Estuary provide unique and important roosting habitat. While long-term monitoring is required to accurately assess the conservation status of Australian insectivorous bats, such as *M. norfolkensis*, the rigorous investigations presented in this thesis deliver essential data to inform current land management practices at both regional and local scales.
References


Kerth, G., N. Perony, and F. Schweitzer. 2011. Bats are able to maintain long-term social relationships despite the high fission-fusion dynamics of their groups. Proceedings of the Royal Society of Biological Sciences 278:2761-2767.


NSW Department of Planning. 2006. Lower Hunter Regional Strategy. Department of Planning, Sydney, NSW.


Appendix 1 - IUCN criteria assessment

Introduction

*Mormopterus norfolkensis* is currently listed as vulnerable C1 under the IUCN red list (IUCN 2009). In light of the new information regarding habitat use and roost selection presented within this thesis, it is timely to re-examine the conservation status of *M. norfolkensis*. To list a particular taxon in any of the categories of threat, only one of the five IUCN listing criteria needs to be met (IUCN Standards and Petitions Subcommittee 2013). However, a taxon should be assessed against as many criteria as available data permit (IUCN Standards and Petitions Subcommittee 2013).

Methods

I estimated the extent of occurrence (EOO) of *M. norfolkensis* by creating a minimum convex polygon from 1062 presence-only database records (Atlas of Living Australia 2013) obtained after filtering to exclude extreme outliers and records before 1990. I used all observation types as *M. norfolkensis* is rarely captured but has a distinctive call able to be identified from ultrasonic recorders. I then used the 4 km$^2$ grid cell method to calculate the area of occupancy (AOO), by tallying the number of grid cells with records and multiplying this by the cell area.

Additionally, in order to provide a conservative estimation of ‘suitable habitat’ within the EOO using a different method, I created a MaxEnt model (Phillips et al. 2006) of all *M. norfolkensis* records and elevation using the Atlas of Living Australia spatial portal (Atlas of Living Australia 2013). I chose elevation as this was the strongest predictor in our landscape habitat models (Chapter 2). I retained 30 % of the records as a testing dataset and assessed discrimination ability using the area under the curve (AUC) of the receiver
operating characteristic. I used the *M. norfolkensis* probability of occurrence threshold of > 0.5 to define suitable habitat. I then calculated the area of suitable habitat within EOO based on the elevation contour specified by the MaxEnt model.

As *M. norfolkensis* was found to have a negative association with urban land-use (Chapters 2, 5 and 6), I next used land-use spatial layers sourced from the Australian Bureau of Statistics to review and predict habitat loss as a result of urbanisation. I calculated the area of urban land-use within the suitable habitat mapped for *M. norfolkensis* in 2006 and also in 2011. Then I calculated the yearly rate of increase and extrapolated this to a 10 year prediction. Finally I assessed the predictions of *M. norfolkensis* occurrence against the IUCN Red List criteria (IUCN Standards and Petitions Subcommittee 2013).

**Results**

I found that *M. norfolkensis* had an EOO of 1,306,596 km² and an AOO of 25,640 km² (Table A1; Figure A1). The MaxEnt model indicated that *M. norfolkensis* was more likely to occur at lower elevations. Specifically, the probability of occurrence threshold > 0.5 equated to elevation < 100 m (Figure A2). The model had good discrimination ability to both the training and test data (AUC > 0.761; Figure A3). Therefore, I defined suitable *M. norfolkensis* habitat as all areas below 100m elevation (Figure A1).
Table A1: IUCN red list criteria description, parameter estimates, methods and reliability ranking for *Mormopterus norfolkensis*

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<td>Predicted 0.99 % loss in suitable habitat over 10 years</td>
<td>No</td>
<td>Indirect measure of projected habitat loss derived from calculations of urban expansion 2006 - 2011 within suitable habitat (defined below)</td>
<td>Low</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>This is a conservative estimate only taking urbanisation impacts into account and only over a narrow timeframe.</td>
<td></td>
</tr>
</tbody>
</table>
| B             | Restricted geographic range (EOO < 20,000 km²; AOO < 2,000 km²) | EOO = 1,306,596 km²  
AOO = 25,640 km²  
Suitable habitat = 316,206 km² | No | EOO: MCP of all records  
AOO: occurrence tally of 4 km² cells within EOO  
Suitable habitat: all area below 100 m elevation minus urban land-use areas | EOO: reliable  
AOO: likely under-estimate  
Suitable habitat: likely over-estimate |
| C             | Small population size (< 10,000 mature individuals) and decline (> 10 % in three generations) | No data | No | No data - cannot be inferred | Low |
| D             | Very small (< 1,000 mature individuals) or restricted population (AOO < 20 km²) | Unlikely < 1,000 mature individuals in total.  
Preliminary mark-recapture estimate from mangrove forest suggest a colony size of 1,320 individuals (McConville unpubl. data). | No | Mark-recapture in mangrove forest applying Lincoln-Peterson index | Moderate.  
Trapping records for the mangrove forest (Chapter 4) also support > 1,000 population size |
| E             | Quantitative analysis (probability of extinction in the wild ≥ 10 % in 100 years) | No data | No | No data | Low |
Figure A1: *Mormopterus norfolkensis* extent of occurrence (EOO) and suitable habitat map derived from a MaxEnt model of occurrence with elevation.

Figure A2: MaxEnt model response of *M. norfolkensis* probability of occurrence against elevation (m).
Based on land-use mapping, I found that the amount of urban land-use increased by 5% between 2006 and 2011, within the area defined as *M. norfolkensis* suitable habitat. However, this is not a 5% loss in habitat. When I removed urban land-use from the area of suitable habitat in both 2006 and 2011 and then predicted the percent reduction of suitable habitat (resulting from urban development) after 10 years, I found less than 1% reduction of suitable *M. norfolkensis* habitat (Table A1). Overall, when I reviewed the various calculations for *M. norfolkensis*, I found that it did not meet any of the IUCN criteria for listing as threatened (Table A1).