CONSERVING AND RESTORING WILDLIFE IN FRAGMENTED URBAN LANDSCAPES: A CASE STUDY FROM BRISBANE, AUSTRALIA




A thesis submitted for the degree of Doctor of Philosophy at The University of Queensland in December 2006

by

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STATEMENT OF ORIGINALITY, CONTRIBUTION TO JOINTLY PUBLISHED WORK, AND CONTRIBUTION OF OTHERS

STATEMENT OF ORIGINALITY

The work presented in this thesis is, to the best of my knowledge and belief, original and my own work, except where otherwise stated in the text. This material has not been submitted, either in whole or in part, for a degree at this or any other university.

CONTRIBUTION TO JOINTLY PUBLISHED WORK

Two jointly published papers, and three jointly prepared papers in review are reproduced in their entirety as chapters forming part of this thesis and my contribution to these was as follows:

Chapter 2. Garden J., McAlpine C., Peterson A., Jones D., Possingham H. (2006) Review of the ecology of Australian urban fauna: A focus on spatially-explicit processes. *Austral Ecology*, 31, 126-148.

- Original idea, literature search and compilation, and all written work.

Chapter 3. Garden J.G., McAlpine C.A., Possingham H.P., Jones D.N. (in review) Using multiple survey methods to detect terrestrial reptiles and mammals: What are the most successful and cost efficient combinations? *Wildlife Research.*

- Original idea, all data collection and analysis, and all written work.

Chapter 4. Garden J.G., McAlpine C.A., Possingham H.P., Jones D.N. (in press) Habitat structure is more important than vegetation composition for local-level management of native terrestrial reptile and small mammal species living in urban remnants: A case study from Brisbane, Australia. *Austral Ecology.*

- Original idea, all data collection and analysis, and all written work.

Chapter 5. Garden J.G., McAlpine C.A., Possingham H.P. (in review) What's more important for wildlife in fragmented urban landscapes – local, patch, or landscape-level influences? A reptile and small mammal case study from southeast Queensland, Australia. *Biological Conservation.*

- Original idea, all data collection and analysis, and all written work.

Chapter 6. Garden J.G., Peterson A., McAlpine C.A., Possingham H.P. (in review) Conserving native terrestrial reptiles and small mammals in urban landscapes: The need for a multi-scaled, multi-species approach to planning and management. *Landscape and Urban Planning.*

- Original idea, all data analysis, and all written work.

CONTRIBUTION OF OTHERS

Dr Clive McAlpine and Prof. Hugh Possingham contributed to the discussion and development of all ideas, and commented on all written material and data analysis. Michiala Bowen and Barbara Triggs provided expert analysis of hair and scat samples, which contributed to species identifications used in the main analyses. The contributions of others to each chapter were:

Chapter 2: Dr Ann Peterson, Assoc. Prof. Darryl Jones and two anonymous reviewers provided comments on the written material.

Chapter 3: Assoc. Prof. Darryl Jones, Dr Nick Clemann, and two anonymous reviewers commented on the written material. Assoc. Prof. Darryl Jones also provided advice on trapping methods during field surveys.

Chapter 4: Dr Grant Wardell-Johnson, Dr Ben Lawson, and Hussein Bashir provided advice and feedback on PATN analysis. Assoc. Prof. Darryl Jones, two anonymous reviewers, and Dr Michael Bull (managing editor of Austral Ecology) commented on the written material.

Chapter 5: Sean Hough conducted FRAGSTATS analysis of data.

Chapter 6: Dr Ann Peterson helped develop ideas and commented on written material.

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Jenni G. Garden Date

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Dr Clive A. McAlpine (Primary Advisor) Date

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Finally, to my family – 'thank-you' seems barely adequate for all you've done for me. I am forever grateful for your love and encouragement. Thank-you for always believing in me and supporting me – I know I will never fall far with you as my safety net.

LIST OF PUBLICATIONS RELEVANT TO THIS THESIS

Garden J., McAlpine C., Peterson A., Jones D., Possingham H. (2006) Review of the ecology of Australian urban fauna: A focus on spatially-explicit processes. *Austral Ecology*, 31, 126-148.

Garden J.G., McAlpine C.A., Possingham H.P., Jones D.N. (in review) Using multiple survey methods to detect terrestrial reptiles and mammals: What are the most successful and cost efficient combinations? *Wildlife Research*.

Garden J.G., McAlpine C.A., Possingham H.P., Jones D.N. (in press) Habitat structure is more important than vegetation composition for local-level management of native terrestrial reptile and small mammal species living in urban remnants: A case study from Brisbane, Australia. *Austral Ecology*.

Garden J.G., McAlpine C.A., Possingham H.P. (in review) What's more important for wildlife in fragmented urban landscapes – local, patch, or landscape-level influences? A reptile and small mammal case study from southeast Queensland, Australia. *Biological Conservation*.

Garden J.G., Peterson A., McAlpine C.A., Possingham H.P. (in review) Conserving native terrestrial reptiles and small mammals in urban landscapes: The need for a multi-scaled, multi-species approach to planning and management. *Landscape and Urban Planning*.

PREFACE

With the exceptions of Chapter 1 (General Introduction) and Chapter 7 (General Discussion), this thesis is presented as a compilation of logically connected published, in press, or in review manuscripts. For the most part, the content of Chapters 2-6 are presented in the format they were submitted or published, with the journal acknowledged at the start of each chapter. Minor stylistic changes have been made for the purposes of maintaining continuity in this thesis. For instance, figures and tables within chapters have been re-labelled to ensure consistency with thesis chapters, 'in press' or 'submitted' references have been updated where possible, and in-text referencing styles have been formatted to suit the thesis. The acknowledgements and references sections for each manuscript have been removed; instead overall acknowledgements for the thesis are provided on pages v-vi, contributions by others to each chapter are shown on pages ii-iv, and a single references list for the thesis is presented at the end of Chapter 7 (pp. 204).

As Chapters 2-6 were written as stand-alone articles for scientific journals, there is some repetition between chapters, particularly in the Introduction, and Materials and Methods sections. Where necessary, permission was obtained to reproduce published papers in this thesis.

At the start of each chapter and the references section, a page of photographic plates is included, with a brief description of each photo. Information for each description was derived from: Queensland Museum (1995), Cogger (2000), Menkhorst and Knight (2001), Wilson and Swan (2003), and Wilson (2005). These plates and their descriptions are not an integral part of the thesis, and are not referred to in the text. Instead, they are included for interest's sake, and as a photographic record of the field surveys and the species detected during this study. All photos were taken by me, except where otherwise acknowledged.

All fauna surveys were conducted under UQ AEC permit GSP/030/04/BCC/UQGS, and Queensland Government EPA scientific purposes permit WISP01975204. Surveys of private properties were conducted with the land-holder's consent.

ABSTRACT

The environmental conditions that make a location suitable for urban development often coincide with those that support high species diversity and endemism. The resulting loss, fragmentation, and degradation of natural habitats have significant ramifications for urban wildlife. Native wildlife populations fragmented by urban development undergo population declines and localised extinctions often long after the development occurs. Native biodiversity is therefore under threat as urban areas continue to expand and replace natural habitats, yet the processes enabling wildlife to persist in urban areas are not well understood. Consequently, urban planning and management decisions often fail to ensure the long-term conservation of urban biodiversity. This project applied a spatially-explicit, multi-scaled landscape approach to determine the relative importance of site, patch, and landscape-level attributes for the occurrence of reptile and small mammal species living in fragmented forest remnants of Brisbane City, Queensland, Australia.

The study tested a set of *a priori* models to investigate the importance of site-level habitat attributes relative to patch size and shape, and landscape composition and configuration for native reptile and small mammal species. Field based fauna and habitat surveys were conducted at 59 sites, with fauna surveys repeated for spring/summer over two consecutive years. Fauna surveys used a combination of live trapping, direct observation, and trace analysis to increase the detection probability of the range of target species. The field surveys provided information about reptile and small mammal species occurrences, and the local-level

habitat structure and composition within each site. Comparative analysis was used to investigate the detection success and costs associated with the different survey methods employed. Cluster analysis and multi-dimensional scaling ordination were used to investigate relationships between species occurrences and local-level habitat characteristics. Generalised linear modelling and hierarchical partitioning were used to determine the importance of the area of forest habitat and its configuration relative to patch size and shape, and local vegetation composition and structure.

A total of 19 reptile and nine mammal species were identified. All survey methods made a contribution to overall detection success by detecting at least one species not identified by any other method. Pit-fall traps and direct observations were the most successful and cost efficient method combination for detecting reptile species. In contrast, mammals were most successfully and efficiently detected using a combination of hair funnels and Elliott traps (for small bodied mammals), or cage traps (for medium sized mammals), with the one exception being the more successful use of pit-fall traps for detecting planigales (*Planigale maculata*).

At the local-level, I found that species composition for both taxa was influenced most by habitat structure rather than vegetation composition. Reptile species composition was correlated with: the amount of fallen woody material, the presence of termite mounds, soil compaction, and the weediness of sites. Mammal species assemblages were most correlated with the presence of grass trees and soil compaction. When the importance of local-level habitat attributes was examined relative to patch and landscape-level attributes, I found that attributes across each spatial level were important for determining species richness. Overall, patch level attributes, such as size and shape, were less important than landscape context and

local attributes of habitat quality, such as habitat structural complexity. Reptile species responded to attributes at both the local and landscape-levels, with the area of forest habitat and its configuration in the surrounding landscape, and soil compaction and weed cover at the local-level, being correlated with species richness. In comparison, mammal species responded to attributes at all three spatial levels. The key factors influencing mammal species were the amount of forest and rural/low density urban habitat at the landscape-level, habitat composition at the local-level, and patch size and shape at the patch-level.

The research outcomes highlight the need to adopt a multi-species, multi-scaled approach to research and urban conservation planning and management. The major outcomes of this project are synthesised into a set of guidelines and a decision-support tree that will enable urban decision-makers to target priority habitat and landscape attributes for the conservation of native reptile and small mammal communities in urban landscapes.

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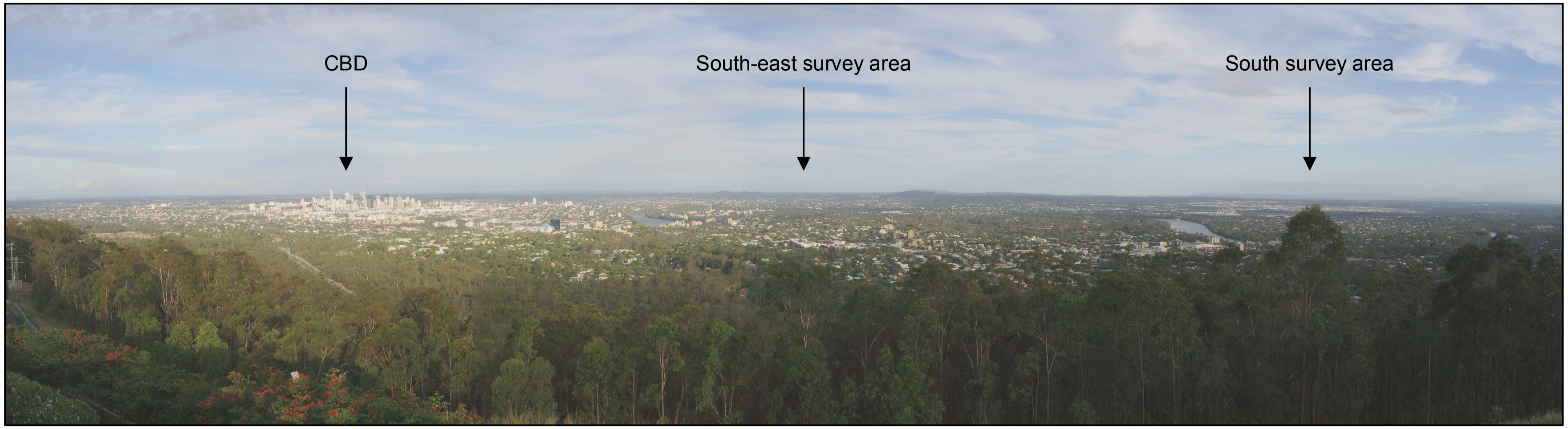


(b)

Plate 1. (a) Brisbane City as viewed from Mt. Coot-tha, with approximate location of south and south-east survey areas indicated; (b) Examples of regional ecosystem (RE) type 12.9-10.4.

CHAPTER 1

GENERAL INTRODUCTION

1.1. BACKGROUND

Urban areas are characterised worldwide by highly-modified environments that have major consequences for native ecosystem composition and dynamics. Whereas endogenous disturbances have long been recognised to benefit native ecosystems, exogenous disturbances resulting from human land-use, fragment and isolate natural habitats to the detriment of many species and ecosystem patterns and processes. The process of urbanisation is amongst the most extreme and rapidly expanding form of global land transformation (Vitousek et al. 1997). It is most pronounced in tropical and subtropical regions, where the rate of human population growth is faster than in any other region of the world (Barrow 1991). At present, almost half the Earth's human population lives in urban environments, with this percentage expected to increase to approximately 60% by 2030 (United Nations 2006).

Such intense modification of native ecosystems produces an environment so different from its original state that even if anthropogenic activities were removed, complete recovery would be highly unlikely (Lugo 2002). Lugo (2002) questions whether we can sustainably manage human-dominated tropical landscapes, concluding that landscapes could be managed in highly developed areas, and that '…deforestation and fragmentation are not unidirectional processes…' (pp. 610). However, Lugo (2002) further notes that for anthropogenic activities to continue, we must accept that the altered environment will be '…a whole new suite of ecosystems with species composition different from those of our natural areas…" (pp. 611). It is therefore crucial that we develop a sound understanding of urbanisation impacts on ecosystem processes and wildlife populations if we are to conserve native biodiversity in urban landscapes.

Urban landscape ecology has attracted attention from numerous disciplines including ecology, sociology, wildlife management, and government sectors and developers. However, despite an increase in disciplinary interest and technological advances (Johnson 1995; Bastin and Thomas 1999; Marzluff and Ewing 2001), there remains a lack of understanding of the effects of urbanisation on ecosystem processes, the behaviour and persistence of wildlife populations, and how these effects vary at multiple spatial scales. Furthermore, the findings from the majority of studies are not effectively integrated into urban land-use planning (Risser 1993, 1996). Consequently, there is a paucity of information available to guide the development and management of urban landscapes for conservation, resulting in continuing uninformed and, often, detrimental conservation decisions for biodiversity.

The key issue then is how to respond to these problems so that impacts on urban biodiversity are minimised and native biodiversity conservation within the urban context is successful. The answer lies in an inter-disciplinary (Johnson 1995; Klopatek and Gardener 1999), hierarchically structured (Cousins 1993; Childress et al. 1999; White et al. 1999), and adaptive approach (Liu and Taylor 2002; Rutledge and Lepczyk 2002). In addition, ecological knowledge must be better integrated into urban decision-making processes; to do so requires ecologists conveying a '…loud clear message to decision-makers…' (Villard 2002, pp. 320).

1.2. PROBLEM STATEMENT

Biodiversity is under threat from accelerating urbanisation, yet the processes enabling wildlife to persist in urban areas are not well understood. Consequently, urban planning and management decisions are often ineffective or incompatible for the long-term conservation of native urban biodiversity. To rectify this problem, urban ecology must investigate multi-scaled influences on a range of native species, particularly those that currently receive little attention. Ecologists must also take steps to integrate ecological findings into planning frameworks and management strategies, by translating priority ecological requirements into a format that may be easily understood and adopted by decision-makers.

For instance, key decisions for conserving urban biodiversity include: should planning and management actions and limited resources (time, money and personnel) be directed towards attributes at the local-level (< 1 ha), patch-level (1 – 100s ha), or landscape-level (100s – 1000s ha)? Furthermore, if for instance patch-level attributes are of greatest importance, what particular attributes are most important? That is, how should actions be prioritised for conserving wildlife species?

1.3. AIMS AND OBJECTIVES

The project aims to disentangle the relative importance of site, patch, and landscape-level variables on the occurrence of native terrestrial reptiles and small mammals living in a fragmented urban landscape of Australia. In particular, the project will address the questions:

(i) Is local-level habitat structure more important than habitat composition?

(ii) How important is the landscape context relative to local-level habitat factors?

(iii) How can ecological information best be integrated into urban landscape planning and management?

The main objectives of this project are to:

(i) Review the Australian urban ecology literature in order to identify gaps in the current knowledge base that should take priority in future research.

(ii) Conduct fauna surveys within lowland remnant vegetation fragments of Brisbane City, using various survey methods, and investigate the success and cost efficiency of combinations of different survey methods.

(iii) Determine the relative importance of local-level habitat structure and habitat composition for influencing reptile and small mammal compositions.

(iv) Investigate the importance of patch and landscape-level influences for reptiles and small mammals, relative to local-level influences.

(v) Develop guidelines for promoting ecologically sensitive urban design and management decisions and actions.

(vi) Propose how biodiversity conservation may be integrated into local, regional, and state planning and management strategies.

1.4. APPROACH

The project applies a hierarchical, spatially-explicit, multi-scale approach to test a set of *a priori* predictive models based on the habitat requirements of multiple species in fragmented urban landscapes. The focal urban landscape was Brisbane, Southeast Queensland, Australia. A combination of live trapping, direct observation, and trace analysis methods were used during fauna surveys to detect the range of target reptile and small mammal species. Local-level habitat characteristics were assessed from habitat surveys conducted at each site. The importance of habitat structure and composition for reptiles and small mammals was investigated using cluster analysis and multi-dimensional scaling ordination techniques. High resolution Quickbird satellite imagery and ArcGIS were used to calculate patch and landscape-level metrics, and hierarchical partitioning and model averaging enabled the relative importance of local, patch, and landscape-level elements to be determined. A set of guidelines and a decision-support tree were then formulated to facilitate the integration of the project's findings into local, regional, and state planning frameworks and management strategies.

1.5. STRUCTURE OF THESIS

With the exception of Chapters 1 and 7, the thesis is presented as a series of logically ordered published, in press, and in review journal papers which present the findings from the project's main objectives (Figure 1.1). Chapter 2 provides a review of the contemporary ecological knowledge regarding Australian urban wildlife. This chapter draws on urban ecology research published between 1990 and 2005, in order to determine gaps in the current knowledge base, and so identify areas of research priority. In Chapter 3, I examine the success and cost efficiency of the various methods used during fauna surveys. Chapter 4 details the local-level attributes that were found to be most important for reptiles and mammals, and Chapter 5 examines the importance of local-level attributes, relative to patch and landscape-level attributes. These findings are then presented in Chapter 6 as a series of guidelines for planning and managing urban landscapes for the benefit of biodiversity. In this chapter I also propose how these guidelines may be integrated into state/regional and local planning frameworks and management strategies. The major findings and management implications from the project are then collated in Chapter 7, where I also outline the project's limitations, and make recommendations for future research.

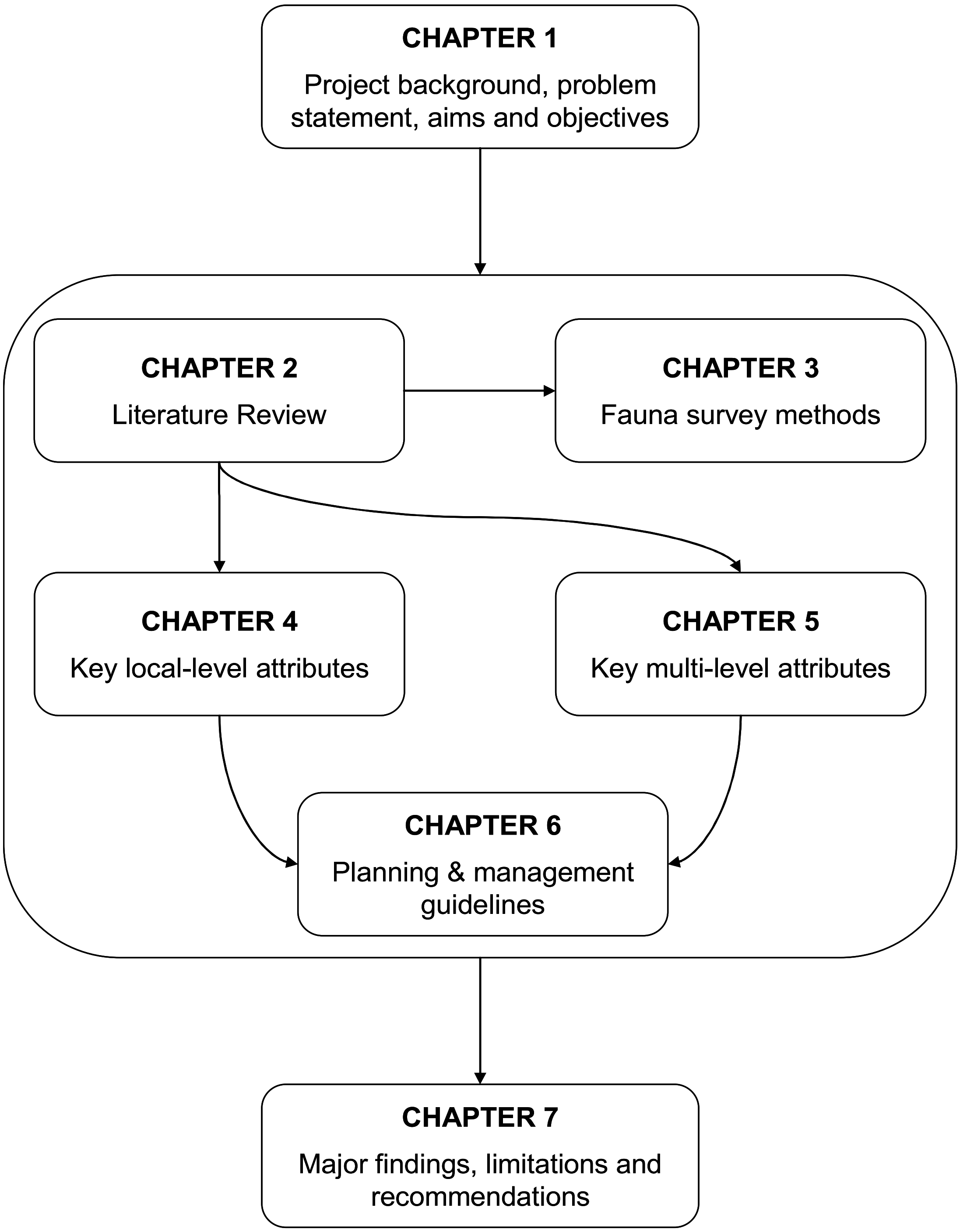


Figure 1.1. Schematic of thesis structure, showing interrelations between chapters.

(a) Yellow-footed antechinus (*Antechinus flavipes*)
- Family: Dasyuridae
- Carnivorous, nocturnal marsupial.
- Size range: HBL 90-160 mm, TL 65-140 mm, 20-75 g.
- Distribution: Queensland, New South Wales, Victoria, South Australia, Western Australia.
- In Brisbane: absent from inner city; uncommon elsewhere, restricted to suitable remnant bushland habitat.

In this photo a male antechinus is being released from observation bag.

(b) Subtropical antechinus (*Antechinus stuartii*)
- Family: Dasyuridae
- Insectivorous, nocturnal marsupial.
- Size range: HBL 93-130 mm, TL 92-120 mm, 18-60 g.
- Distribution: Queensland, New South Wales.
- In Brisbane: common where specific suitable remnant forest habitat occurs.

Photo from: http://www.abc.net.au/goulburnmurray/stories/s1439616.htm

(c) Common dunnart (*Sminthopsis murina*)
- Family: Dasyuridae
- Insectivorous, nocturnal marsupial.
- Size range: HBL 65-100 mm, TL 68-90 mm, 12-28 g.
- Distribution: Queensland, New South Wales, Victoria, South Australia.
- In Brisbane: uncommon, restricted to suitable remnant bushland habitat in peri-urban landscapes.

In this photo, the individual's tail tip has been temporarily marked white

(d) Common planigale (*Planigale maculata*)
- Family: Dasyuridae
- Insectivorous, nocturnal marsupial.
- Size range: HBL 70-95 mm, TL 60-90 mm, 6-12 g.
- Distribution: Queensland, New South Wales, Northern Territory, Western Australia.
- In Brisbane: restricted to, but reasonably common in, suitable remnant bushland habitat.

In this photo, the rim of a pit-fall bucket is visible behind the planigale.

Plate 2. Dasyurids: (a) Yellow-footed antechinus; (b) Subtropical antechinus; (c) Common dunnart; (d) Common planigale.

Chapter 2

REVIEW OF THE ECOLOGY OF AUSTRALIAN URBAN FAUNA: A FOCUS ON SPATIALLY-EXPLICIT PROCESSES

Citation: Garden J., McAlpine C., Peterson A., Jones D., Possingham H. (2006) Review of the ecology of Australian urban fauna: A focus on spatially-explicit processes. Austral Ecology, 31, 126-148.

2.1. ABSTRACT

Cities have a major impact on Australian landscapes, especially in coastal regions, to the detriment on native biodiversity. Areas suitable for urban development often coincide with those areas that support high levels of species diversity and endemism. However, there is a paucity of reliable information available to guide urban conservation planning and management, especially regarding the trade-off between investing in protecting and restoring habitat at the landscape level, and investing in programs to maintain the condition of remnant vegetation at the local (site) level. We review the literature on Australian urban ecology, focusing on urban terrestrial and aquatic vertebrate and invertebrate fauna. We identify four main factors limiting our knowledge of urban fauna: 1) a lack of studies focusing at multiple ecological levels; 2) a lack of multi-species studies; 3) an almost total absence of long-term (temporal) studies; and, 4) a need for stronger integration of research outcomes into urban conservation planning and management. We present a set of key principles for the development of a spatially-explicit, long-term approach to urban fauna research. This requires an understanding of the importance of local-level habitat quality and condition relative to the composition, configuration, and connectivity of habitats within the larger urban landscape. These principles will ultimately strengthen urban fauna management and conservation planning by enabling us to prioritise and allocate limited financial resources to maximise the conservation return.

Keywords: Australia, habitat condition, ecological level, government decision-making, landscape structure, urban fauna.

'The future is not just what lies ahead; it is something that we create.'

(Forman and Collinge 1997, p.129)

2.2. INTRODUCTION

The primary driver of the global decline of biodiversity is habitat loss and fragmentation resulting from anthropogenic pressures on natural ecosystems. Urbanisation is arguably the most damaging, persistent and rapidly expanding form of anthropogenic pressure (Vitousek et al. 1997; Lugo 2002; McKinney 2002; Miller and Hobbs 2002). Almost half (49.2%) the world's population currently resides in urban areas (United Nations 2006). As the human population continues to increase, so too do demands for residential, industrial, commercial, and recreational space. Consequently, the Earth's landscapes are becoming increasingly urbanised; by 2010 humans will be a predominantly urban species, with 51.3% predicted to be living in urban areas (United Nations 2006).

The primary impacts of urban development on biodiversity are extensive habitat loss and fragmentation, which significantly alters the structure of urban landscapes, and the composition and structure of ecosystems embedded in these landscapes (Forman 1995; Baskin 1998; Wilcove et al. 1998; Marzluff and Ewing 2001; Faulkner 2004). Consequently, urban landscapes are dominated by the built environment composed of buildings, bridges, roads, and paved areas, interspersed with 'green' habitat patches ranging from cultivated parks and gardens to remnant bushland, all of which vary in size, shape, and condition (Burgman and Lindenmayer et al. 1998; Angold et al. 2001). The cumulative impacts of urban development on fauna species are not restricted to urban areas, but may extend varying distances into neighbouring landscapes and affect species and ecological processes in adjacent ecosystems (Quarles et al. 1974; Rees 1997; Bisonnette 2002; Yeoman and Mac Nally 2005). As this urban footprint continues to expand and intensify, remnant vegetation patches and their dependent biota are subject to further habitat loss, fragmentation, and degradation (Fahrig 1997, 2001). Such intense and widespread modification of natural landscapes and ecosystems produces an environment so different from its natural state that even if anthropogenic activities were removed, complete recovery would be unlikely (Lugo 2002).

Despite the significant destruction and degradation of habitats, urban areas have the capacity to support a wide diversity of vertebrate and invertebrate fauna species, perhaps due to the range of diverse natural and artificial habitat niches and conditions that occur in urban areas (Niemelä 1999a, 1999b; Collins et al. 2000). Kühn et al. (2004), however, argues that high species diversity in urban areas occurs not because of but in spite of urbanisation, with urban development often coinciding with areas of naturally high species diversity and endemism, such as coastal and tropical regions. This diversity of fauna species exhibits varying responses to urbanisation. The magnitude and direction of urban impacts on each species depends on that species life-history attributes, sensitivities to environmental disturbances, interspecies interactions, and dispersal ability (Dickman and Doncaster 1989; Cox et al. 2003; Tischendorf et al. 2003). This diversity of responses has previously been used to categorise species based on similarities in their responses to, and abilities to persist within, the urban environment (e.g., Baskin 1998; McKinney 2002; Catterall 2004). For the purposes of this paper, species are discussed as either: 'matrix-occupying', 'matrix-sensitive', or 'urban-sensitive'. Matrix-occupying species are those that commonly dominate the urban matrix due to their ability to move through and live within the built matrix. Conversely, matrix-sensitive species perceive the built matrix as: unsuitable habitat with a lack of food and shelter resources, a barrier to movement, and an area of increased risk of predation. Consequently, matrix-sensitive species are often restricted to vegetation patches of suitable habitat, resulting in fragmentation of populations and increasing the risk of localised extinctions. Species classed as urban-sensitive are unable to persist in urban landscapes, even in remnant patches of native vegetation. Characteristics shared by these species include limited dispersal abilities, and narrow or specialised dietary requirements

Prior to the 1990s, urban areas were largely overlooked or ignored in ecological studies as they were considered to be non-viable habitat for fauna populations and therefore of no use for conservation efforts (Botkin and Beveridge 1997; McDonnell et al. 1997; Savard et al. 2000). As a result, the impacts of urbanisation on fauna populations are not well understood, often resulting in poorly targeted conservation actions (Niemelä 1999a; Recher 2002). The results of studies conducted in non-urban environments (e.g., forest, agricultural) are not necessarily transferable to urban areas because of the increased rate and complexity of environmental changes, coupled with additional urban-based pressures such as increased densities of transport networks and associated usage, domestic dog and cat predation, and significantly altered abiotic factors (e.g., water, noise, air, and soil pollution). Therefore management actions based on knowledge accrued from non-urban research may be inappropriate within the urban environment. This lack of appropriate ecological knowledge hinders the long-term success of existing and proposed urban conservation actions. This is of particular concern for tropical and coastal regions that commonly support high levels of biodiversity, yet where the rate of urban development is also higher than in any other region (Barrow 1991; Kühn et al. 2004).

Such issues are particularly evident in Australia, where the capital cities and most major urban developments are located along the nation's seaboard, especially the eastern seaboard (Commonwealth of Australia 2003a). Australia's human population is already predominantly urban, with more than 90% residing in urban areas (United Nations 2006). The fraction of people living in cities continues to increase, with the highest rate of recent urban development occurring along the eastern tropical and subtropical coast, which also supports high species diversity and endemism (Queensland Museum 1995; Commonwealth of Australia 2003b).

The plight of biodiversity in Australia's urban landscapes has been recognised by Adams (1994), Jones (2003), and Lunney and Burgin (2004). To minimise this loss, urban ecological researchers, both internationally and in Australia, need to understand and predict species-habitat relationships at multiple ecological levels, and determine the relative importance of the amount of habitat, its spatial configuration, and its condition on species survival. This knowledge is a prerequisite for urban planners and conservation managers to effectively conserve and restore urban biodiversity.

This paper provides a constructive review of the scientific literature of Australian-based urban ecological research in order to highlight the strengths and limitations of existing research knowledge. We focus on terrestrial vertebrate and invertebrate fauna, with an emphasis on spatial processes and long-term trends. Some studies on aquatic fauna are also included. Our primary target audience is urban ecologists, but we also highlight the need for research to meet the information needs of local government and regional planning authorities responsible for urban biodiversity conservation and whom also fund some ecological research. The paper is divided into three sections. First, we outline criteria used to select the papers reviewed, with a focus on recent Australian urban fauna studies. Next, existing research strengths and gaps are identified. Third, we develop a set of key principles to address these gaps and guide the development of a spatially-explicit, long-term approach to the study of Australia's urban fauna, and thereby improve the conservation of Australia's urban fauna.

2.3. REVIEW CRITERIA

The research papers reviewed (Table 2.1) were selected based on the following criteria:

1. Focus on fauna (vertebrate and invertebrate)-habitat relationships within Australian urban areas:

For the purposes of this review, urban areas are defined as 'areas of intense human influence, dominated by the built environment and supporting a population cluster of more than 1000 people'. This definition is based on previous definitions by Forman and Godron (1986), Pickett et al. (2001), and the Australian Bureau of Statistics (Commonwealth of Australia 2003a). Papers were selected if they addressed fauna (vertebrate and invertebrate, terrestrial and aquatic) habitat relationships within Australian urban areas. We acknowledge the relevance of other studies of urban ecology issues such as: native and exotic pest species management (e.g., Marks and Bloomfield 1999; Matthews et al. 2004; Moriarty 2004; Ross 2004), human-fauna interactions and conflicts (e.g., Jones and Everding 1991; Miller et al. 1999; Shine and Koenig 2001; Warne and Jones 2003), changes in species behaviour, diet, and fecundity (e.g., Smith and Carlile 1993; Statham and Statham 1997; Webster et al. 1999; Fearn et al. 2001; Parry-Jones and Augee 2001; Rollinson and Jones 2002; Hoye and Spence 2004; Markus and Hall 2004; Temby 2004; Everding and Jones 2006), wildlife mortalities due to vehicle collisions and predation (e.g., Barratt 1997; Koenig et al. 2002; Dique et al. 2003; Taylor and Goldingay 2004), urbanisation impacts on flora species (e.g., Rose 1997; Buist et al. 2000; Leishman et al. 2004; Stenhouse 2004), and abiotic influences such as air, light, and water pollution on urban flora and fauna populations (e.g., McDonnell et al. 1993; Riley and Banks 1996; Angold 1997; Spooner et al. 2003).

2. Published between 1990-2005:

The surge of urban-based fauna ecology studies over the last 10-15 years influenced the decision to review only those articles published between 1990 and 2005. We acknowledge that several urban fauna-habitat studies were conducted in Australia prior to 1990, with the majority of these focussing on avian communities, particularly matrix-occupying assemblages (e.g., Jones 1981; Ford 1983; Jones 1983; Green 1984; Mason 1985; Catterall et al. 1989; Munyenyembe et al. 1989). However, their exclusion is not expected to significantly impact on the review content as many 'modern-day' studies draw and expand on the findings of earlier studies.

3. Keyword searches and accessibility:

We selected publications in refereed journals, books, and conference proceedings that were accessible via online databases and library searches. Online databases were searched using a combination of the following keywords: urban, suburban, city, town, wildlife, fauna, animal/s, mammal, bird, avian, reptile, amphibian, herpetofauna, aquatic, invertebrate, insect, coast/al, fragmentation, ecology, habitat, site, patch, landscape/s, scale, level, environment, mosaic, development, Australia. An examination of citations within selected papers was used to identify further references. Where possible, if a relevant paper was not available via online databases or library searches, authors were contacted and a copy of the paper requested.

Table 2.1. Summary of 63 Australian urban ecology studies conducted between 1990 and 2005. Studies are grouped according to: birds, mammals, herpetofauna (reptiles and amphibians), invertebrates, and aquatic. Ecological levels are assigned according to definitions provided in Table 2.2. Studies that were not spatially-explicit are classified as: PVA (Population Viability Analysis), long-term (field study repeated over $\geq$ 10 years), historical (comparison of historical records to present species distribution patterns), or review (of current knowledge base). † Indicates studies of multiple taxa that are repeated in each relevant section of this table, with key findings discussed relative to each specific taxon.

Species	Ecological Level/s	Location	Key Findings and Habitat Features	Reference
BIRDS				
Splendid fairy-wren *Malurus s. splendens*	PVA	(near) Perth, WA	– Habitat loss coupled with fire frequency. – Habitat fragmentation. – Secondary pressures such as nest predation.	Brooker and Brooker (1994)
Powerful owl *Ninox strenua*	Site	Melbourne, VIC	– Requires structurally diverse vegetation.	Cooke et al. (2002)
Australian Magpie *Gymnorhina tibicen*	Long-term	Perth, WA	– Habitat loss – prefers cleared, well watered areas such as urban lawns and open parks. – Distributional changes over time.	Wood and Recher (2004)
Rainbow Lorikeets *Trichoglossus haematodus* Musk Lorikeets *Glossopsitta concinna*	Site	Melbourne, VIC	– Habitat type (streetscapes preferred). – Vegetation composition (native species preferred). – Vegetation age (well established streetscapes preferred).	Fitzsimons et al. (2003)
Willie wagtail *Rhipidura leucophrys* Pied currawong *Strepera graculina*	Site	Various	– Vegetation density. – Inter-species interactions (predation).	Major et al. (1996)
Multiple owl species	Review	Sydney, NSW	– Habitat loss and fragmentation – extensive bushland areas must be conserved. – Prey species habitat area must also be conserved.	Kavanagh (2004)

Multiple nectarivorous species	Site	Sydney, NSW	– Vegetation composition (native plants more important than exotics).	French et al. (2005)
Multiple species	Site Patch, Landscape	Southeast QLD (3100 km^2)	– Overall relative importance: spatial, then habitat characteristics. – Specifically specialist species: connectivity and patch shape. – Specifically migrants and generalist species: site characteristics.	Bentley and Catterall (1997)
Multiple species	Patch, Landscape	Brisbane, QLD	– Patch size, vegetation abundance and composition, spatial configuration of habitats. – Inter-species interactions.	Catterall (2004)
Multiple species	Patch, Landscape	Brisbane, QLD	– Vegetation structure. – Habitat loss and alteration.	Catterall et al. (1998)
Multiple species	Site, Patch	Brisbane, QLD	– Patch area. – Habitat heterogeneity. – Vegetation structure and density (particularly understorey density).	Grover and Slater (1994)
Multiple species	Long-term	Townsville, QLD	– No changes in overall species richness. – Significant changes in species composition. – Changes in species distributions attributed to changes in vegetation structure and composition over time (species-specific preferences) and associated inter-species interactions.	Jones and Wieneke (2000)
Multiple species	Patch	Sydney, NSW	– Habitat heterogeneity.	Parsons et al. (2003)
Multiple species	Site	Sydney, NSW	– Vegetation composition. – Inter-species interactions.	Parsons and Major (2004)
Multiple species	Site, Patch, Landscape	Perth, WA	– Habitat complexity and vegetation structure – native plants, not exotics. – Patch isolation and habitat fragmentation. – Urban matrix and resource availability within suburbs. – Fire management, vehicular mortality and predation	Recher (2004)

Multiple species	Patch	Perth, WA	– Vegetation structure – maintain complex understorey. – Ground cover. – Resource availability from residential gardens.	Recher and Serventy (1991)
Multiple species	Patch	Brisbane, QLD	– Vegetation structure – retain old growth canopy trees. – Vegetation composition – remnant vegetation more important than revegetated patches.	Sewell and Catterall (1998)
Multiple species	Site	Goode Beach, WA	– Vegetation composition and structure. – Inter-species interactions.	Smith (2002)
Multiple species	Site	Melbourne, VIC	– Vegetation composition and structure. – Habitat condition (development disturbance).	White et al. (2005)
Multiple species	Site, Landscape	Wollongong, NSW	– Wetland pollution. – Habitat heterogeneity – especially maintain wetlands. – Vegetation structure and heterogeneity. – Habitat connectivity and buffers. – Disturbances from human recreational activities and pets.	Wood (1993)
Multiple species	Site, Patch, Landscape	Melbourne and Geelong, VIC	– Patch size. – Density of urban development.	Yeoman and Mac Nally (2005)
Multiple species †	Site, Patch	Perth, WA	– Small, isolated patches (1ha) important for urban birds – including migratory species. – Vegetation composition (native remnants vegetation better than exotic plantings). – Species-specific responses.	Cooper (1995)
Multiple species †	Patch, Landscape	Sydney, NSW	– Patch area (and edge effects): ~ 4 ha threshold for generalist/urban tolerant species and ~ 50 ha for fragmentation sensitive species. – Functional connectivity between habitat patches.	Drinnan (2005)
Multiple species †	Historical	Perth, WA	– Compositional changes since European settlement. – Combination of habitat loss and fragmentation, altered fire regimes, introduced exotic species, wetland modification.	How and Dell (1993)

Multiple species †	Historical	Adelaide, SA	– Species composition varies significantly over time. – Habitat loss and fragmentation. – Vegetation composition and structure. – Edge effects. – Inter-species interactions. – Species-specific responses.	Tait et al. (2005)
MAMMALS				
Long-nosed bandicoot *Perameles nasuta*	PVA	Sydney, NSW	– Urbanisation threats to adult mortality more immediately important than managing habitat availability (although effects are additive).	Banks (2004)
Long-nosed bandicoot *Perameles nasuta*	Site	North Head, NSW	– Vegetation structural complexity and habitat heterogeneity.	Chambers and Dickman (2002)
Long-nosed bandicoot *Perameles nasuta*	Site	Sydney, NSW	– Vegetation structural complexity and habitat heterogeneity. – Mortalities due to vehicle collisions and predation by exotic species.	Scott et al. (1999)
Eastern barred bandicoot *Peremeles gunnii*	Site	Hamilton, VIC	– Vegetation structural complexity and habitat heterogeneity. – Utilise both natural and artificial materials for shelter.	Dufty (1994)
Northern brown bandicoot *Isoodon macrourus*	Site, Patch, Landscape	Brisbane, QLD	– Functional connectivity between habitat patches. – Patch area. – Habitat quality (especially ground cover density).	FitzGibbon et al. (submitted)
Koala *Phascolarctos cinereus*	Landscape	Koala Coast, QLD	– Habitat loss and fragmentation. – Habitat connectivity essential, especially between large patches. – Connectivity between urban remnants and ex-urban remnant bushland.	Dique et al. (2004)
Koala *Phascolarctos cinereus*	Patch, Landscape	Noosa Shire, QLD	– Habitat loss and habitat quality. – Patch area. – Density of sealed roads.	McAlpine et al. (2005)
Koala *Phascolarctos cinereus*	Site, Patch, Landscape	QLD and NSW	– Combination of environmental variables across three spatial levels. – Presence and abundance of food trees most important overall.	McAlpine et al. (2006a)

Koala *Phascolarctos cinereus*	Landscape	Port Stephens, NSW	– Remnant habitat spatial configuration and connectivity. – Road density. – Fire frequency and intensity. – Relative importance of these factors varies spatially.	Rhodes et al. (2006)
Koala *Phascolarctos cinereus*	Patch, Landscape	Noosa, QLD	– Habitat loss and fragmentation.	Seabrook et al. (2003)
Koala *Phascolarctos cinereus*	Site, Patch, Landscape	Warringah Shire, NSW	– Habitat loss. – Patch connectivity. – Habitat quality, especially food tree density. – Management of mortalities due to pet predation and vehicle collisions. – Mitigation of dispersal barriers such as fences and walls.	Smith and Smith (1990)
Koala *Phascolarctos cinereus*	Site, Landscape	Sydney, NSW	– Habitat loss and connectivity. – Vegetation composition and substrate. – Fire regimes, vehicle and dog mortalities and, weed invasions.	Ward and Close (2004)
Platypus *Ornithorhynchus anatinus*	Site	Melbourne, VIC	– Utilise both natural waterways and drainage channels. – Burrow location influenced by physical attributes of banks (concave profile preferred over convex profiles) and associated riparian vegetation density and structure.	Serena et al. (1998)
White-striped freetail bat *Tadarida australis*	Site	Brisbane, QLD	– Hollow bearing tree characteristics (e.g., height, diameter, senescence). – Surrounding local vegetation (i.e. tree density and undergrowth).	Rhodes and Wardell-Johnson (2006)
Squirrel glider *Petaurus norfolcensis*	PVA	Brisbane, QLD	– Habitat loss. – Patch isolation and area. – Functional connectivity between suitable patches. – Habitat quality (both patches and connecting vegetation corridors). – Road density.	Goldingay and Sharpe (2004)
Squirrel glider *Petaurus norfolcensis*	Site, Patch, Landscape	Brisbane, QLD	– Vegetation composition and structure. – Remnant habitat area and altitude. – Habitat isolation and connectivity.	Rowston et al. (2002)

Various (gliders and possums)	Site	Wyong and Lake Macquarie, NSW	– Vegetation structure, composition and age. – Fire frequency and inter-species interactions. – Species-specific responses.	Smith and Murray (2003)
Multiple species	Historical	Melbourne, VIC	– Species-specific responses. – Habitat loss and fragmentation. – Patch connectivity and habitat quality. – Predation by introduced species.	van der Ree (2004)
Multiple species †	Historical	Adelaide, SA	– Habitat loss and fragmentation. – Vegetation structure. – Inter-species interactions. – Species responses vary. – Species composition varies significantly over time.	Tait et al. (2005)
Multiple species †	Historical	Perth, WA	– Combination of habitat loss and fragmentation, altered fire regimes, introduced exotic species, wetland modification.	How and Dell (1993)
Multiple species †	Patch, Landscape	Perth, WA	– Habitat fragmentation.	How and Dell (2000)
Multiple species †	Site, Patch	Perth, WA	– No native mammals found in study patch (1ha) although representatives from other groups were.	Cooper (1995)
HERPETOFAUNA				
Estuarine crocodile *Crocodylus porosus*	Site	Northeast coastline, QLD	– Riparian vegetation clearing anthropogenic developments. – Secondary impacts: motor boat disturbance, commercial netting and individual removals.	Kofron and Smith (2001)
Multiple skink species	Site, Patch	Cumberland Plain, NSW	– Edge effects influence species distributions within patch.	Anderson and Burgin (2002)

Multiple lizard species	Site, Patch, Landscape	Hobart, TAS	– Vegetation structure and composition (native plants better than exotics). – Patch geology and aspect. – 1 species influenced by patch size; none by habitat fragmentation.	Jellinek et al. (2004)
Multiple reptile species †	Historical	Perth, WA	– Combination of: habitat loss and fragmentation, local level habitat degradation, altered fire regimes, and wetland modification. – Species-specific responses.	How and Dell (2000)
Multiple reptile species	Patch	Perth, WA	– Combination of large and small remnants important. – Large remnants especially important for urban snakes. – Species-specific responses.	How and Dell (1994)
Multiple reptile species †	Patch, Landscape	Perth, WA	– Patch area important for all reptiles except skinks. – Small remnants (as small as 1 ha) important for reptiles – although need to manage for fire and predator exclusion. – Annual variation in lizard assemblages. – Species-specific responses.	How and Dell (1993)
Multiple amphibian species †	Patch, Landscape	Sydney, NSW	– Patch area: thresholds at ~ 4 ha for generalist/urban tolerant species and ~ 50 ha for intolerant species. – Edge effects. – Functional connectivity between habitat patches.	Drinnan (2005)
Multiple herpetofauna species †	Site, Patch	Perth, WA	– Small, isolated patches (1ha) – no large reptiles although historically present in the area; but important for small reptiles and amphibians. – Remnant habitat.	Cooper (1995)
Multiple herpetofauna species †	Historical	Adelaide, SA	– Reptile species composition (not richness) varies significantly over time. – Substrate type and vegetation structure. – No native amphibian losses since European settlement. – Species-specific responses.	Tait et al. (2005)

Multiple herpetofauna species	Historical	Sydney, NSW	– Decrease in species richness for both reptiles and amphibians. – Large reptiles (e.g., goannas) locally extinct in urban remnant patches. – Amphibians influenced by degradation of water quality. – Arboreal amphibians more negatively impacted than terrestrials. – Species-specific responses. – Secondary impacts such as predation, altered fire regimes and human intervention influence both reptile and amphibian species.	White and Burgin (2004)
INVETEBRATES				
Ghost crab *Ocypode cordimana*	Site	Sydney, NSW	– Fewer burrows on urban beaches than non-urban beaches. – Human activity and dune modification affects species distribution – not yet clearly understood.	Barros (2001)
Western Jewel butterfly *Hypochrysops halyaetus*	Site	Perth, WA	– Vegetation composition and structure. – Habitat condition – prefer degraded, post-fire habitats. – Inter-species interactions (mutual ant partner).	Dover and Rowlingson (2005)
African rhinoceros beetle *Temnorhynchus retusus*	Site	Sydney, NSW	– Exotic vegetation has supported successful invasion and establishment.	Krell and Hangay (1998)
Multiple arthropod species	Site	Perth, WA	– Tree age. – Tree species (native better than exotics).	Bhullar and Majer (2000)
Multiple arthropod species	Site, Patch, Landscape	Sydney, NSW	– Habitat fragmentation. – Patch size. – Habitat condition and altered fire regimes. – Proximity to urban matrix. – Species-specific responses.	Gibb and Hochuli (2002)
Multiple ant species	Site	Brisbane, QLD	– Habitat heterogeneity. – Inter-species interactions.	Burwell and Grimbacher (2005)

Multiple snail species	Review	Sydney, NSW	– Small remnant vegetation patches as important as large ones. – Vegetation type, habitat loss and fragmentation and habitat condition. – Species-specific responses.	Clark (2004)
Multiple insect species	Site	Sydney, NSW	– Vegetation diversity.	Emery and Emery (2004)
Multiple insect species	Patch	Sydney, NSW	– Patch size. – Inter-species interactions.	Hochuli et al. (2004)
Multiple butterfly species	Review	Various	– Vegetation composition (especially exotic weeds). – Inappropriate fire regimes.	New and Sands (2002)
AQUATIC				
Multiple estuarine fish species	Site, Landscape	Botany Bay, NSW	– Anthropogenic activities significantly alter fish habitats. – Able to use significantly modified urban estuarine habitats in the absence of dispersal barriers (connectivity).	Gibbs (2004)
Multiple epilithic diatoms	Landscape	Melbourne, VIC	– Water quality (electrical conductivity) and drainage connections.	Newall and Walsh (2005)
Multiple amphipod species	Landscape	Melbourne, VIC	– Urban density and stormwater drainage connections. – Sealed roads and associated run-off also threaten species survival.	Walsh et al. (2004)
Multiple macrofauna species	Landscape	Sydney, NSW	– Adjacent habitat type and land use.	Yerman and Ross (2004)

2.4. REVIEW OF AUSTRALIAN URBAN FAUNA RESEARCH

The review was stratified based on five broad fauna groupings: birds, mammals, herpetofauna (reptiles and amphibians), invertebrates, and aquatics. With the exception of the aquatic group, all studies discuss either terrestrial or arboreal species, including species that utilise both aquatic and terrestrial systems (e.g., platypus, crocodiles, frogs and crabs). Aquatic studies include both freshwater and oceanic species, and microscopic organisms. Table 2.1 segregates studies based on these fauna groupings, and indicates the ecological level/s at which studies were conducted, as well as the location and key findings of each study. We use the term 'level' to refer to levels of ecological organisation, as opposed to the term 'scale', which refers to spatial resolution and extent of the analysis (*sensu* Turner et al. 2001). Three levels are identified: site or *in situ* (< 1 ha), patch (1-100s ha) and landscape (100s -1000s ha). Examples of environmental characteristics operating at each level are provided in Table 2.2.

2.4.1. Birds

Birds are by far the most obvious and readily identifiable fauna element of urban habitats and, as such, have received more research attention than other animal groups (35% of all papers, Figure 2.1a). Australian urban bird assemblages are generally species rich, yet individual local communities may be characterised by either low species richness with high abundance or, high species richness with low abundance of individual species (Wood 1993). The increase in abundance of some bird species is attributed to their ability to utilise plentiful and novel resources (e.g., food and nesting locations) and habitats that occur in human-modified environments (Jones and Wieneke 2000; Catterall 2004). Such species usually comprise matrix-occupying assemblages, which dominate the built urban matrix and are comprised of both introduced species and native species (Jones and Wieneke 2000; Catterall 2004; Tait et al. 2005). These matrix-occupying assemblages are dominated by behaviourally aggressive, medium-bodied and large-bodied species such as, Australian magpies (*Gymnorhina tibicen*), butcherbirds (*Cracticus spp*.), currawongs (*Strepera spp*.) noisy miners (*Manorina melanocephala*), and lorikeets (*Trichoglossus spp*.) (Jones and Wieneke 2000; Fitzsimons et al. 2003; Jones 2003; Wood and Recher 2004; Catterall 2004). This pattern of domination differs from the general trend observed in cities of the Northern hemisphere where exotic and small-bodied natives tend to dominate the urban matrix (*sensu* Jones and Wieneke 2000). Much of the recent ecological research conducted on matrix-occupying species has investigated diet and behavioural adaptations to the urban environment, as well as human-avian conflict management issues (Jones and Everding 1991; Smith and Carlile 1993; Major et al. 1996; Fulton and Ford 2001; Hasebe and Franklin 2003; Ross 2004).

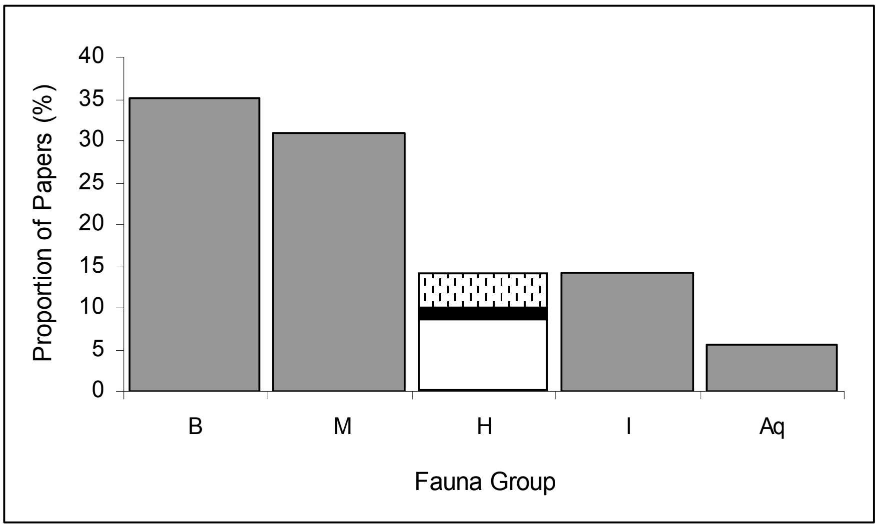
Some research has also investigated the influence of habitat on the distribution and abundance of Australian avian matrix-occupiers, showing that despite their dominance within the built environment, avian matrix-occupiers are rarely ubiquitous across the urban matrix, with distributions being influenced by *in-situ* habitat factors such as vegetation composition and structure. Lorikeets, for example, appear to prefer well established streetscapes planted with flowering native tree species (Fitzsimons et al. 2003), while Australian magpies thrive in highly disturbed, vegetated areas with little canopy cover such as well-watered residential lawns, managed parks and sporting ovals (Wood and Recher 2004). Noisy miners, in contrast, tend to dominate moderately disturbed ‘edge’ habitats with neither dense nor predominantly cleared vegetation cover (e.g., Catterall 2004; Tait et al. 2005).

Table 2.2. Examples of habitat characteristics operating at three ecological levels.

Ecological Level	Environmental Characteristics
Site/Local (<1 ha)	• Vegetation composition and structure • Ground cover type and proportion • Soil compaction • Nutrient levels
Patch (1-100s ha)	• Size and shape • Perimeter:area ratio (edge effects) • Distance to landscape features (e.g., patch, river, road) • Time since isolation
Landscape (>100 ha)	• Total habitat area (habitat loss) • Number of habitat patches (habitat fragmentation) • Degree of connectivity between patches • Density of land-use types (e.g., roads, residential areas, parks)

In situ habitat factors have also been shown to be important for determining the presence and distribution of matrix-sensitive species, which in Australia are often small-bodied, insectivorous and nectarivorous bird species (White et al. 2005). Most studies that have investigated *in situ* habitat relationships agree that conserving, and potentially increasing, urban avian richness in habitat patches is primarily dependent on creating and maintaining structurally complex and floristically diverse habitats, with native plant species being recommended over exotic species (e.g., Recher and Serventy 1991; Cooke et al. 2002; Parsons et al. 2003; French et al. 2005; White et al. 2005). Sewell and Catterall (1998) further recommended that the retention of remnant vegetation habitats (more so than revegetation of native species) was central to the successful recovery and maintenance of insectivorous bird species in urban areas.

(a)



(b)

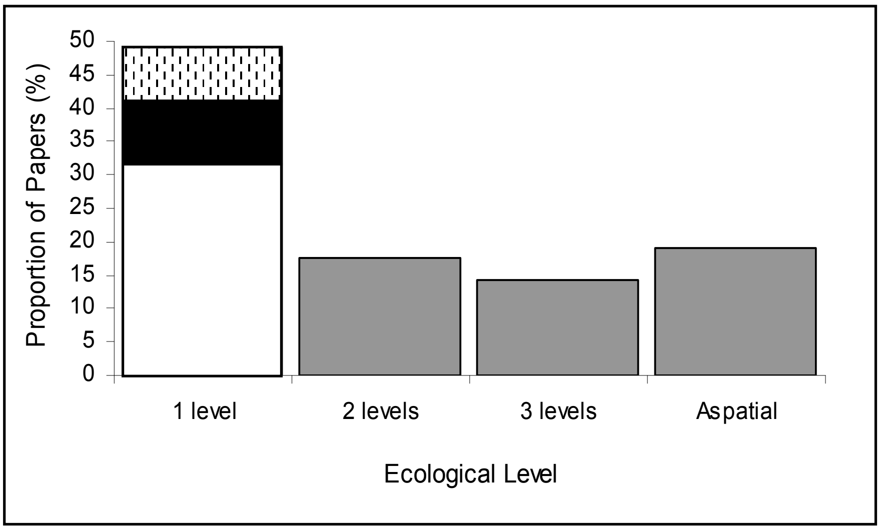


Figure 2.1. Proportion of fauna groups and ecological levels examined across 63 urban fauna studies (see Table 2.1). (a) Relative proportion of fauna groups: B=birds; M=mammals; H=herpetofauna; I=invertebrates; Aq=aquatic. Herpetofauna column shows the relative number of studies that examined: reptiles (white area), amphibians (black area), and both reptiles and amphibians (dashed area). (b) Relative proportion of ecological levels (see Table 2.2): one level, two levels, three levels or, aspatial (no explicit level examined). The single level column shows the relative number of studies that examined: site level (white area), patch level (black area) and landscape level (dashed area). Aspatial includes PVA, long-term, historical and review studies as indicated in Table 2.1.

Both patch and landscape level variables have also been found to influence the presence of birds within urban landscapes. Kavanagh (2004) recommended that the conservation of various urban owl species necessitates the retention and protection of large areas of contiguous bushland. Similarly, Yeoman and Mac Nally (2005) concluded that patch size, rather than vegetation structure, influenced the avifauna of coastal moonah (*Melaleuca lanceolata*) woodlands in the Melbourne and Geelong regions of Victoria, although the density of urban development also appeared to affect bird species richness. Similarly, other studies have reported the influence of various combinations of multi-level habitat factors such as, patch size and area, spatial configuration, connectivity and isolation of patches, patch context and habitat heterogeneity and also, *in-situ* vegetation structure and composition (e.g., Wood 1993; Cooper 1995; Catterall 2004; Recher 2004; Drinnan 2005). The relative influence of these factors, however, appears to vary between species. Bentley and Catterall (1997), for instance, found that both spatial and habitat characteristics influenced avian assemblages overall, although spatial characteristics such as fragmentation, isolation, and the matrix were especially important for habitat-specialist species, whereas, migrant and habitat-generalist species were influenced more by site characteristics such as the structural complexity of vegetation (see also: Wood 1993; Cooper 1995; Catterall 2004; Drinnan 2005; Tait et al. 2005).

The influence of interspecies interactions, associated with environmental variables, is also a contributing factor to species distribution and abundance across urban areas. For example, Major et al. (1996) found that pied currawong (*Strepera graculina*) populations increased significantly as vegetation density decreased, which in turn increased their ability to predate the nests of willie wagtails (*Rhipidura leucophrys*), reducing the long-term local viability of willie wagtail populations in urban areas. Smith (2002) suggested that in Goode Beach, Western Australia, red wattlebirds (*Anthochaera carunculata*) became the dominant large honeyeater species in areas where vegetation structure and composition was suitable for their nesting requirements. This domination coupled with their inherent aggressive behaviour contributed to the disappearance of smaller bodied bird species from the same areas. Catterall (2004) reported invasions of flocks of behaviourally aggressive noisy miners adjacent to moderately disturbed forest edges excluded smaller bodied birds from these areas, despite the presence of suitable habitats for such species (see also: Low 2002; Parsons and Major 2004).

The cumulative effect of species-specific responses to other species and environmental variables results in urban avifauna assemblages that differ significantly from non-urban and pre-urban assemblages. What is more, as urban areas are in a continual state of flux (habitat destruction, degradation, and restoration), so too are the associated avian assemblages. As the time since development increases, avian assemblages tend to become more distinct from pre-urban and non-urban assemblages, yet more similar to those in other urban environments (Jones and Wieneke 2000; Chace and Walsh 2006). Studies of such long-term changes in Australian urban avifauna assemblages are less common in the literature, with notable exceptions occurring in Perth (e.g., Recher and Serventy 1991; Smith 2002; Recher 2004; Wood and Recher 2004), Adelaide (Tait et al. 2005), and Townsville (Jones and Wieneke 2000). Given the restricted number of studies with a long-term focus, coupled with the inherent differences between sites in terms of biophysical and climatic influences, it is unclear whether observed trends are consistent across different urban environments (Jones and Wieneke 2000). This lack of historical knowledge, coupled with species-specific responses to urban environmental change, makes it particularly problematic to predict changes in avifauna populations living within Australia's expanding urban environments.

2.4.2. Mammals

Native mammals, particularly medium-sized species, are widely considered in Australia to be the fauna group most detrimentally affected by habitat changes resulting from urbanisation and associated secondary impacts such as introduced species predation, interspecies competition, and mortality from vehicle collisions (How and Dell 1993; Tait et al. 2005). Very few mammal species become human commensals, with exceptions of exotic species, such as the house mouse (*Mus musculus*), rabbit (*Oryctolagus cuniculus*), and red fox (*Vulpes vulpes*), and the native brushtail possum (*Trichosurus vulpecula*). To effectively conserve rich and viable urban mammal assemblages it is essential that species-habitat relationships are clearly understood for a range of species with different habitat requirements and dispersal abilities. To date, much of the research conducted on urban mammalian assemblages has examined various aspects of species diet and behaviour (e.g., Markus and Hall 2004), with significantly fewer studies specifically investigating multi-level habitat relationships within the dynamic urban environment.

The most commonly studied mammal species are those that are easily observed and identified, as well as those that elicit an emotive public response. Accordingly, possum and glider species (e.g., Smith and Murray 2003; Goldingay and Sharpe 2004; Matthews et al. 2004), koalas *Phascolarctos cinereus* (e.g., Lunney et al. 2002; Dique et al. 2004; Ward and Close 2004; McAlpine et al. 2005), and bandicoot species (e.g., Dufty 1994; Chambers and Dickman

2002; Banks 2004) are the most frequently studied Australian urban mammals. More cryptic, rare or survey intensive species, such as dunnarts (*Sminthopsis spp.*), antechinus (*Antechinus spp.*), native rodents (e.g., *Rattus fuscipes*, *Xeromys myoides*), and volant species (bats) have received significantly less research focus. This bias is a particular concern for many of Australia's native marsupials and rodents, which are often both behaviourally cryptic, and unrecognised or disliked by the public.

Terrestrial mammal species and, to a lesser extent, gliding species, have a lower probability of survival in urban environments due to their limited locomotion and dispersal abilities which make them either unable or unwilling to traverse the urban matrix and transport networks between often highly fragmented and isolated remnant patches (Dickman and Doncaster 1987, 1989; How and Dell 1993; Goldingay and Sharpe 2004; Tait et al. 2005). Individuals dispersing between remnant vegetation patches face anthropogenic barriers such as walls, fences, and roads, as well as increased risk of predation from exotic species (e.g., cats *Felis catus*, dogs *Canis lupis familiaris*, red foxes *Vulpes vulpes*), or collisions with vehicles (Andrews 1990; Smith and Smith 1990; Barratt 1997; Forman 1999; Lunney et al. 2002; Dique et al. 2003; Banks 2004).

As with avifauna, attaining a comprehensive understanding of the habitat relationships of mammal assemblages in urban areas is particularly challenging. This challenge stems from the diverse range of habitat and disturbance factors, operating at multiple ecological levels, which impact species distribution, abundance, and species-specific responses to habitat change and predation pressures. How and Dell (2000), for example, suggested that mammals in general are most influenced by habitat fragmentation yet, McAlpine et al. (2005) concluded habitat loss is more important than habitat fragmentation and road density for koala populations living in semiurban landscapes of southeast Queensland. Within their home range, though, koalas prefer particular Eucalyptus species, especially those occurring on fertile soils (e.g., Smith and Smith 1990; Moore et al. 2004; Ward and Close 2004). Bandicoots, however, prefer spatially heterogenous and structurally complex habitats to support foraging and shelter requirements (e.g., Dufty 1994; Scott et al. 1999; Chambers and Dickman 2002). For northern brown bandicoots (*Isoodon macrourus*) living in Brisbane, Queensland, FitzGibbon et al. (submitted), showed the combined influence of patch level characteristics such as patch size and functional connectivity between patches at the landscape level were important. Functional connectivity was identified as the most important factor for conservation actions targeting bandicoot populations. For glider species living in Brisbane, however, it is recommended that mitigating the impacts of habitat loss and fragmentation at the landscape-level is the highest management priority, followed by habitat condition and edge effects at the local level (Goldingay and Sharpe 2004).

Rhodes et al. (2006) has further demonstrated that the relative importance of environmental factors may also vary for individual species, depending on the level of observation. They found that habitat loss and fragmentation, connectivity, patch size, vegetation composition and road density all influenced urban koala population dynamics in Port Stephens Shire, New South Wales. However, at the site level, tree species composition was the primary factor, while the amount of habitat and its connectivity were important at the landscape-level. Such knowledge has implications for targeting fauna management actions and determining the levels of government responsible for these actions (D. Lunney pers. comm., 2004).

Determining effective management strategies for urban mammal populations is further complicated by the lack of knowledge regarding population trends over time. Very few studies have explicitly considered population trends and viability, whilst those that have tried to address this problem have been limited by the consistency of historical records (e.g., How and Dell 1993; van der Ree 2004; Tait et al. 2005). To achieve effective long-term conservation of mammals in urban areas, it is essential that urban researchers broaden their current focus to include multiple species; especially those currently underrepresented in the scientific literature, and determine the influence and relative importance of environmental variables across multiple ecological levels. These factors must also be examined within a temporal context in order to facilitate accurate predictions of population responses to ongoing urban development and associated environmental change.

2.4.3. Reptiles and Amphibians

Reptiles and amphibians may be collectively referred to as herpetofauna, although the taxa exhibit significantly different responses to urbanisation and associated environmental changes. Of the papers reviewed, 14% focussed on urban herpetofauna species, with reptile species examined more frequently than amphibians (Figure 2.1a). This may be a consequence of more reptile species (than amphibians) being easily observed in urban landscapes, the cryptic nature of many amphibians, and the concomitant difficulties associated with amphibian surveys (Hazell 2003). At present, there are major knowledge gaps in the understanding of the habitat requirements of both reptile and, particularly, amphibian populations within urban environments. For instance, it is apparent that certain herpetofauna species are able to adapt to urban environments, whilst others are restricted to remnant vegetation patches or disappear locally. However, this pattern of persistence and loss appears to vary regionally between urban environments. Large reptile species, particularly snakes, have been repeatedly reported as the most detrimentally impacted reptile species in both Perth and Sydney (e.g., How and Dell 1994; Cooper 1995; How and Dell 2000). Yet, Fearn et al. (2001), documented the abundance of carpet pythons (*Morelia spilota*) in highly disturbed urban areas, reporting on the removal of 258 'nuisance' pythons from Brisbane and Ipswich (Queensland) over a six year period. Both reptile and amphibian assemblages in Sydney have experienced losses of species since European settlement (White and Burgin 2004), yet Tait et al. (2005) reported only reptile, not amphibian, species have declined in Adelaide despite significant and widespread habitat alterations and water quality degradation.

As with birds and mammals, there is a paucity of information regarding the relative importance of habitat factors operating at multiple ecological levels for herpetofauna assemblages. It appears that the relative importance of habitat factors varies between reptiles and amphibians, as well as between species of each group (e.g., How and Dell 2000; Anderson and Burgin 2002; Jellinek et al. 2004). For example, according to Jellinek et al. (2004) lizards were influenced more by *in situ* vegetation structure and composition rather than patch size (except for one species). Whereas Drinnan (2005) found patch size was especially influential for amphibian species. White and Burgin (2004), however, suggest that frogs are impacted primarily by changes in water cycling and quality. Drinnan (2005) further suggested that habitat thresholds existed for amphibian species, with threatened and urban-sensitive species requiring significantly larger habitat patches (> 50 ha) for survival than matrix-sensitive species that occupied much smaller patches (~ 4 ha).

The current knowledge base clearly lacks a sounds grasp of the relative importance of habitat factors measured at a range of ecological levels and the presence of critical habitat retention thresholds for individual reptile and amphibian species and assemblages. There is also a need to specifically examine and compare temporal variations in herpetofauna assemblages within and among urban environments across Australia. Management actions based on the current limited knowledge base may fail to address critical environmental factors and so fall short in achieving their long-term conservation outcomes. Clearly, much more research is required to better understand both reptile and amphibian population dynamics in urban areas and how these dynamics vary across multiple ecological levels.

2.4.4. Invertebrates

Invertebrates are recognised by the scientific community as being the largest and most diverse fauna group, referred to as ‘the other 99%’ (Ponder and Lunney 1999; Stanisic 2005). However**,** they are one of the least understood groups, both in non-urban and urban landscapes. Because invertebrates play a significant role in the functioning of ecosystems as decomposers, parasites, pollinators and prey for many higher order species (Bhullar and Majer 2000), it is essential that the impacts of urbanisation on invertebrates are explicitly understood. A synthesis of current research knowledge shows that both small and large remnant vegetation patches (Gibb and Hochuli 2002; Clark 2004; Hochuli et al. 2004), vegetation diversity (Emery and Emery 2004; Burwell and Grimbacher 2005), and habitat condition (Gibb and Hochuli 2002; New and Sands 2002; Dover and Rowlingson 2005) are critical factors for invertebrate conservation. Other factors important for individual species include: altered fire regimes (Gibb and Hochuli 2002; New and Sands 2002; Dover and Rowlingson 2005) and interspecies interactions (Hochuli et al. 2004; Burwell and Grimbacher 2005; Dover and Rowlingson 2005). Hochuli et al. (2004), for example, documented that invertebrate herbivore composition varied with remnant patch size, with abundance increasing in smaller patches because of a lower density of predatory and parasitic species that are sensitive to habitat fragmentation. This interaction, in turn, resulted in smaller native vegetation patches, in general, experiencing increased insect herbivory compared to large patches where predatory and parasitic species act to maintain herbivorous insect populations at a sustainable level. The resulting loss of habitat condition is likely to have further implications for the compositions of higher order species such as birds, mammals and reptiles that are particularly sensitive to changes in habitat condition.

As for other fauna groups, the impacts of urbanisation on invertebrate communities are highly species-specific. For example, New and Sands (2002) recommended that the conservation of urban butterfly species required improving the condition of degraded urban habitats, specifically by addressing weed and fire issues. Dover and Rowlingson (2005), conversely, reported that improving habitat condition would negatively influence the already threatened western jewel butterfly (*Hypochrysops halyaetus*), as this species appeared to prefer degraded habitats, particularly those degraded by fires. Subsequently, it was suggested that this particular butterfly species may benefit from urbanisation, and the commonly associated degraded habitats, as long as the species mutual ant partner is also present (Dover and Rowlingson 2005). Similar interspecies interactions have been observed for other invertebrates, although this is sometimes due to species exclusion rather than species mutualism. Heterick et al. (2000), for example, concluded that the composition of native ant species in suburban Perth gardens varied inversely to the presence of exotic ant species, with few native species able to co-exist in gardens dominated by introduced ant populations. Burwell and Grimbacher (2005) reported similar impacts on native ant assemblages at one site in Brisbane as the result of habitat domination by the introduced coastal brown ant (*Pheidole megacephala*).

Temporal variations in invertebrate assemblages have received significantly less examination, although Bhullar and Majer (2000) reported that older, native (especially locally endemic) trees lining Perth's streets hosted both increased abundance and diversity of arthropod species, compared to recently planted native and exotic species. However, the underlying mechanisms driving these variable arthropod responses remain unclear. Such knowledge has important implications for local government management decisions, especially considering the flow-on effect of invertebrate presence influencing invertebrate predators at higher trophic levels. None of the other invertebrate studies reviewed considered temporal dimensions and so the influence of time on invertebrate species dynamics remains unclear.

It is obvious that immediate attention must be paid to both investigating and comprehending the impacts of urbanisation on insect communities, and the concomitant interspecies relationships and invertebrate impacts on the condition of remnant patches and the population dynamics of other invertebrate and vertebrate species. By understanding such relationships, urban planners and conservation managers will be better able to make conservation decisions that target and benefit a range of species and species groups.

2.4.5. Aquatic

Aquatic ecosystems and assemblages are often overlooked in urban ecology studies in favour of the more dominant terrestrial systems, despite the overall concession by researchers that aquatic ecosystems are undeniably impacted by urbanisation. Aquatic fauna have been shown to be particularly sensitive to: altered environmental flows, increased water pollution, turbidity and sediment levels, exotic species introductions, altered vegetation presence and composition (e.g., sea grasses and mangroves), and built structures such as piers, harbours, and dams (e.g., Hough 1995; Dow and Dewalle 2000; Gibbs 2004).

Similar to their terrestrial counterparts, certain urban aquatic species are able to adapt and persist in the modified environments better than others. Gibbs (2004), for instance, concluded that many fish species were able to utilise significantly modified urban estuarine habitats, provided other factors such as water flow, are effectively managed. Comparatively, Newall and Walsh (2005) reported that the built environment, particularly urban stormwater drain designs, had significant negative impacts on diatom assemblages. Walsh et al. (2004) reported similar influences of urban development and stormwater drainage connections, on the distribution of an already threatened stream-dwelling amphipod (*Austrogammarus australia*). Walsh et al. (2004) further commented on the role of sealed roads in urban areas in degrading amphipod habitat (water quality) by altering storm water run-off and transporting urban pollutants into waterways. Yerman and Ross (2004) examined the influence of different landscape types and uses on the composition of mangrove macrofauna communities in Sydney, concluding that mangrove forests adjacent to natural salt marshes support higher macrofauna diversity than do mangroves situated next to man-made parks or bund walls.

The limited number of studies on aquatic fauna-habitat relationships (Figure 2.1a) highlights a paucity of information on aquatic ecosystems. The current knowledge base provides limited, often suppositional, insights into urbanisation impacts on aquatic habitats and the associated responses of their fauna inhabitants. An important initial step for addressing this issue is to alter the way in which urban researchers, public members and urban managers, alike, view aquatic ecosystems and fauna. There is a need to integrate them into the urban ecosystems/fauna viewpoint, rather than treating them as a separate entity.

2.5. PRINCIPLES

Urbanisation and its impacts on fauna populations must be closely examined and understood before urban planners and managers can hope to successfully achieve long-term conservation goals in urban areas. Currently, Australia's conservation actions often fall short of their intended long-term goals due to decision-making processes that are ill-informed by sound scientific research. There are numerous gaps in Australia's urban ecology knowledge base, with certain groups and species being considerably less understood than others. To address these issues we suggest the following five guiding principles when designing and conducting research into the ecology of Australia's urban fauna. By doing so, future research in this field will produce a more comprehensive knowledge foundation upon which to base conservation and management decisions and thereby help urban planners and managers make better decisions.

Principle 1: Urban ecology studies need to adopt a hierarchical landscape approach that explicitly considers the structure of the urban landscape and the influence of the quality and quantity of habitat elements that constitute that landscape.

The urban environment is a complex mosaic of landscape elements dominated by the built environment and interspersed with remnants of natural ecosystems and open space. Remnant ecosystem patches provide habitat for many native fauna species, and vary with respect to shape, size, condition, connectivity/isolation and disturbance regime. Urban landscapes are in a continual, and often rapid, state of flux as a result of human land use pressures. Accordingly, urban fauna populations that inhabit these landscapes are dynamic, and sensitive to changing environmental conditions. Of the research papers reviewed, 49% focussed on the impact of environmental factors operating at a single ecological level, multi-level studies were examined in 32% of cases, and the remaining 19% were aspatial (Figure 2.1b). Of the single level studies, more than half (65%) were conducted at the local level (Figure 2.1b). This bias limits our ability to determine the ecological level that most influences populations, which in turn limits our ability to make recommendations about priority and cost-effective actions for conservation.

Increasingly, urban researchers are demonstrating that urban fauna respond to a combination of habitat variables occurring at multiple ecological levels (e.g., How and Dell 1993; Grover and Slater 1994; Gibb and Hochuli 2002; Rowston et al. 2002; Catterall 2004; McAlpine et al. 2006a). Consequently, studies conducted at a single ecological level (e.g., Dufty 1994; Fitzsimons et al. 2003; Hochuli et al. 2004; Yerman and Ross 2004) can explain only part of the overall impact of urbanisation and may indeed obscure or exaggerate regional declines that are now occurring for many species (Wiens 1994; Hobbs 1999). This does not mean that single-level habitat attributes are unimportant, rather that it is necessary to consider the structure of the whole landscape and the inherent hierarchy of habitat influences when predicting the effects of urbanisation on native biota and choosing conservation actions (McGarigal and McComb 1995; Hokit et al. 1999; Dorner et al. 2002; McAlpine and Eyre 2002). Studies should also recognise that the perceived permeability of the urban matrix differs between species and is not always considered unsuitable (Opdam et al. 2003). Studies that classify urban landscapes as binary (i.e. suitable habitat or unsuitable habitat) risk misinterpreting the influence of urban landscape structure on species occurrence and abundance, and their interactions with other organisms. It is imperative that urban ecology researchers adopt a spatially-explicit approach that specifically considers the scale of movement of the target species, and how it perceives and utilises the dynamic habitat heterogeneity of urban landscapes (Pearson et al. 1996; White et al. 1999; Hostetler and Holling 2000; Debinski et al. 2001; McAlpine et al. 2002).

Principle 2: Urban fauna studies should explicitly test *a priori* predictions of the relative importance of habitat amount, configuration and condition, the presence of critical habitat retention thresholds, and the interaction between these factors.

Critical issues for many Australian local government agencies responsible for urban biodiversity conservation are: how much habitat is enough to sustain viable fauna populations?, how should this habitat be spatially arranged in the landscape? and, what is the relative importance of habitat amount, configuration and condition? These questions are important for prioritising investment in the conservation and restoration of urban biodiversity. Therefore, understanding the relative importance of these factors for various fauna species (terrestrial and aquatic, vertebrate and invertebrate) is essential for informing the decision-making process and investment prioritisation. For example, in Brisbane City, a common dilemma for local government urban planners and conservation managers is whether to focus efforts and resources on restoring the quality of habitats (condition), or on increasing the area of habitat and its connectivity (S. McLean, pers. comm., 2003). This decision is often specific to a particular species, assemblage or landscape/ecosystem in a particular region. If a landscape has experienced extensive habitat loss, and the remaining habitat is below a critical threshold, then increasing the amount of habitat and its connectivity may deliver the greatest conservation outcome. Conversely, if the remaining habitat within the urban landscape is above a critical threshold and is located in a few large patches, then managing habitat quality for species may prove more beneficial for achieving long-term conservation goals. These management issues are further complicated in urban landscapes by the effect of roads, human disturbance pressures (e.g., vandalism, high fire frequencies) and the high density of exotic predator species.

Improving the effectiveness of urban conservation strategies therefore is reliant on being able to prioritise the importance of the amount of habitat, its configuration and condition, whilst minimising the negative impacts of roads, predators and human activity. The hypothetical curve depicted in Figure 2.2 represents how knowledge about the relative importance of environmental variables could be used to assist the decision-making process prioritising conservation actions and investments to ensure effective biodiversity outcomes. The effectiveness of such investments firstly depends on urban researchers explicitly testing the relative importance of, and interactions between, these critical habitat factors.

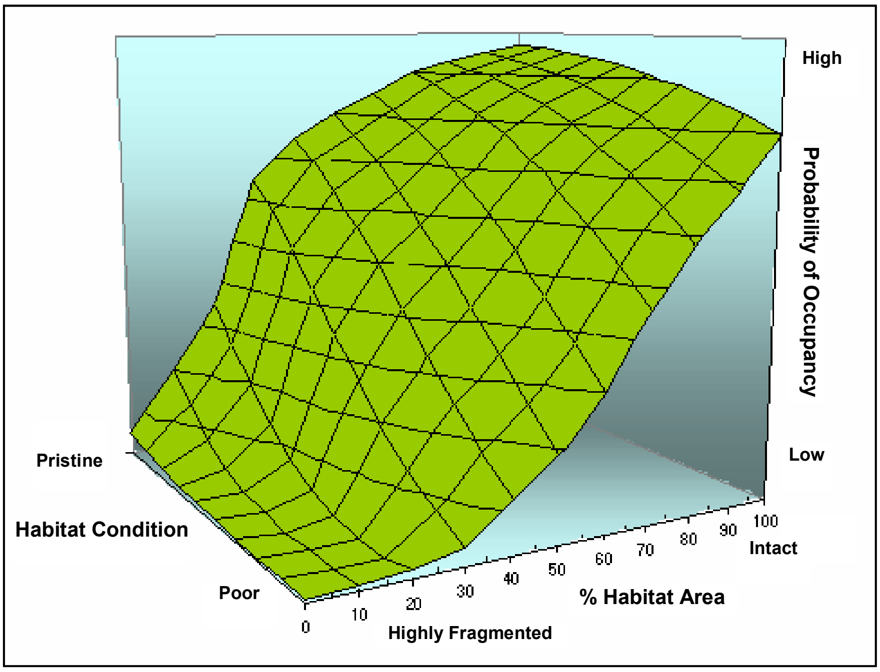


Figure 2.2. Hypothetical curve of the relationships between habitat area, habitat condition (quality), and the probability of a species being present.

Principle 3: Urban ecology studies need to consider the responses of multiple species to urban habitat conditions and dynamics.

Current urban biodiversity management decisions are too often based on recommendations accrued from studies of a single species or a small group of species. However, species have varying habitat requirements and sensitivities to urban landscape change (Table 2.1). A single/few species approach, therefore, is limited in its ability to conserve the full complement of species in the landscape. If maintaining species diversity is the ultimate goal, then landscapes should not be planned and managed based on the requirements of a single or limited selection of species. Often research resources are only available to survey a single species or taxa (e.g., birds). The findings of these studies need to be integrated with similar designed studies in order to develop an integrated, multi-species approach to urban biodiversity conservation and restoration.

An alternative approach considers multiple species ecological ‘profiles’ that classify a range of different species based on similarities in their responses to landscape structure and change (Opdam et al. 2002). The development of comprehensive ecological profiles is a major, yet necessary, challenge that will facilitate urban planning and conservation management decisions that effectively target the habitat requirements of several species and taxa, rather than those specific to a single species or taxon. Such an approach is complicated but particularly pertinent in urban areas because of fauna population fluctuations in response to the extreme, dynamic and continually changing urban environment.

Principle 4: Urban ecological studies need to consider the temporal dimension as well as the spatial dimension of urban landscapes.

Urban landscapes have a temporal as well as a spatial component (*sensu* Marucci 2000) Following clearing or other major disturbances, the remaining habitats often experience decline in condition to a state of lower habitat quality. The response of fauna populations may also occur over long time periods, with certain species exhibiting a time lag of decades or centuries in their responses (Tilman et al. 1994; Hanski 1998; Possingham and Field 2001; Tait et al. 2005). In Australia’s urban landscapes, it is particularly important to study the long-term dynamics of fauna populations in order to predict long-term population responses to urban developments and plan to mitigate the impacts of associated habitat alterations and secondary pressures. For example, many Australian bird and mammal species are long-lived and consequently, non-viable populations may persist for many years in the urban landscape before suddenly disappearing. Yet, very few of the studies reviewed explicitly considered the temporal aspect. Static studies cannot determine population viability and may mistakenly consider a population as being viable. As a consequence, valuable conservation resources may be invested inappropriately. Further, changes in species compositions and distributions across urban habitats, in response to urban land use changes, may also be more accurately predicted using temporal patterns. Jones and Wieneke (2000), for example, demonstrate distinct compositional and distributional shifts in Townsville bird communities over a 16 year period in response to habitat succession and modification, such as transitions from open grassy areas, to recently vegetated areas, to well established areas, and vice versa. Understanding the mechanisms by which species to respond to given disturbances is important for predicting the long-term outcome for urban fauna populations. Incorporating a temporal dimension in urban ecology research will help produce information that will enable urban planners and managers to make effective decisions regarding the impact of prospective development actions on the long-term viability of urban fauna populations.

Principle 5: Urban ecological research must be effectively communicated to urban planners and conservation managers so that recommendations are adopted and integrated into urban planning, management, conservation, and restoration strategies. Within Australia, urban conservation goals are limited by a lack of collaborative research and management (*sensu* Underwood 1995; Liu and Taylor 2002). In order to preserve urban biodiversity, planners, managers and developers must be equipped with appropriate scientific knowledge on which to base sound conservation decisions and actions. Such decisions are often resource (time and money) limited. A critical goal for researchers is to provide 'decision tools' and planning guidelines that facilitate the prioritisation of conservation actions as well as the identification of cost efficient management alternatives. Furthermore, urban ecology researchers should aim to address priority issues for urban managers and planners and take the responsibility for clearly communicating their findings and recommendations in a format that is easily understood by decision makers. In addition, when the multiple-use of habitats is mandated, the successful preservation and restoration of urban habitats for biodiversity conservation rely on a cooperative, multi-disciplinary approach (Johnson 1995). As such, researchers are further encouraged to instigate collaborative projects that incorporate a variety of disciplines (e.g., sociology, ecotourism) and key stakeholders (e.g., landholders, politicians, architects and developers) (e.g., Niemelä 1999a; Collins et al. 2000; Grimm et al. 2000; Musacchio and Wu 2004). Doing so will promote the integration of scientific recommendations into decision-making processes and subsequent planning, management, conservation, and restorations strategies.

2.6. CONCLUSIONS

The process of urbanisation causes significant and ongoing alterations to natural landscape compositions and ecosystem processes. It is axiomatic that such habitat alterations impact the composite fauna communities and inhibit long-term persistence. Ironically, it seems the inherent dynamic nature of cities creates and maintains a complex mosaic of habitat niches, many of which are unique to urban areas, which in turn support a high diversity of fauna species, including rare and threatened species (Niemelä 1999a, 1999b). The future conservation of Australia's urban fauna depends heavily on collaborative research projects that integrate multiple ecological levels of habitat influence and multiple species, as well as temporal variation, in investigating the relationships existing between urban environments and their resident fauna populations. The principles outlined in this paper provide the scientific basis for designing urban fauna studies to address the right questions and help meet the information needs of urban planners and conservation mangers. Without such knowledge, poorly targeted urban fauna conservation actions will continue and populations will decline further under increasing development and disturbance pressures, resulting in a reduced quality of life for all city dwellers.

(a) Bush rat (*Rattus fuscipes*)

- Family: Muridae
- Selective omnivore, nocturnal rodent.
- Size range: HBL 100-205 mm, TL 100-195 mm, 50-225 g.
- Distribution: Queensland, New South Wales, Victoria, South Australia, Western Australia.
- In Brisbane: common in remnant bushland habitat in peri-urban landscapes.

Photo from: http://www.communitywebs.org/ScientificExpeditionGroup/pages/MinPic.htm

(b) Swamp rat (*Rattus lutreolus*)

- Family: Muridae
- Opportunistic omnivore, diurnal and nocturnal rodent.
- Size range: HBL 120-200 mm, TL 56-147 mm, 55-160 g.
- Distribution:.Queensland, New South Wales, Victoria, Tasmania, South Australia.
- In Brisbane: uncommon and restricted to specialised remnant habitat in peri-urban landscapes.

Photo from: http://www.communitywebs.org/ScientificExpeditionGroup/pages/MinPic.htm

(c) Northern brown bandicoot (*Isoodon macrourus*)

- Family: Peramelidae
- Omnivorous, nocturnal marsupial.
- Size range: HBL 300-470 mm, TL 80-210 mm, 500-3000 g.
- Distribution: Queensland, New South Wales, Northern Territory, Western Australia.
- In Brisbane: common in inner and outer suburbs, where suitable habitat is available.

This photo shows me holding a male northern brown bandicoot.

(d) Long-nosed bandicoot (*Perameles nasuta*)

- Family: Peramelidae
- Omnivorous, nocturnal marsupial.
- Size range: HBL 310-450 mm, TL 120-150 mm, 850-1100 g.
- Distribution: Queensland, New South Wales, Victoria.
- In Brisbane: restricted to suitable remnant bushland habitat in peri-urban landscapes.

Photo from: http://filin.vn.ua/mammels/bandikut.htm

Plate 3. Native rodents and bandicoots: (a) Bush rat; (b) Swamp rat; (c) Northern brown bandicoot; (d) Long-nosed bandicoot.

Chapter 3

USING MULTIPLE SURVEY METHODS TO DETECT TERRESTRIAL REPTILES AND MAMMALS: WHAT ARE THE MOST SUCCESSFUL AND COST EFFICIENT COMBINATIONS?

Citation: Garden J.G., McAlpine C.A., Possingham H.P., Jones D.N. (in review) Using multiple survey methods to detect terrestrial reptiles and mammals: What are the most successful and cost efficient combinations? Wildlife Research.

3.1. ABSTRACT

The selection of methods for wildlife surveys is a decision that will influence the accuracy and comprehensiveness of survey outcomes. The choice of methods is commonly based on the species of interest, yet is often limited by the project budget. Although several studies have investigated the effectiveness of various survey techniques for detecting terrestrial mammal and reptile species, none have provided a quantitative analysis of the costs associated with different methods. We compare the detection success and cost efficiency of cage traps, Elliott traps, pit-fall traps, hair funnels, direct observation, and scat detection/analysis for detecting the occurrence of terrestrial reptile and small mammal species in urban bushland remnants of Brisbane City, Queensland. These analyses are tangential to a larger research project and so the findings are a guide only. Cage traps and Elliott traps coupled with hair funnels were the most cost-effective methods for detecting the highest number of ground-dwelling mammal species. Pit-fall traps and direct observations were the most cost-effective methods for maximising the number of reptile species identified. All methods made a contribution to overall detection success by detecting at least one species not detected by any other method. This suggests a combination of at least two complementary methods will provide the most successful and cost-efficient detection of reptile and mammal species in urban forest remnants. Future studies should explicitly test these findings and examine efficient trapping combinations across different habitat types and for other species groups.

Key words: Cage trap, Elliott trap, pit-fall trap, hair funnel, direct observation, scat analysis, Brisbane.

3.2. INTRODUCTION

Managing Australia's terrestrial habitats for mammal and reptile conservation requires a thorough knowledge of the composition and distribution of species within and across the habitats of interest. Wildlife surveys of species occurrence and abundance have long been used to acquire such knowledge. For terrestrial mammal and reptile species, a variety of survey methods have been used by researchers to determine occurrence and abundance. These methods have been specifically designed to target particular species or groups of species

(Sutherland 1996a; Menkhorst and Knight 2001), and so vary in their applicability and relative detection success for different taxa. In general, the choice of survey method(s) is a critical factor influencing the accuracy and comprehensiveness of survey results.

Several studies have investigated the relative success of different survey methods for detecting mammal and/or reptile species in Australian landscapes including, but not limited to: pit-fall traps with or without drift fences (e.g., Mengak and Guynn 1987; Friend et al. 1989; Laurance 1992; Catling et al. 1997; Crosswhite et al. 1999; Moseby and Read 2001; Ryan et al. 2002), Elliott traps (e.g., Laurance 1992; Catling et al. 1997; Clemann et al. 2005), wire cage traps (e.g., Friend 1978; Laurance 1992; Catling et al. 1997), direct observations and/or active searches (e.g., Brown and Nicholls 1993; Catling et al. 1997; Crosswhite et al. 1999; Ryan et al. 2002), hair tubes or funnels (e.g., Catling et al. 1997; Lindenmayer et al. 1999; Mills et al. 2002), and vocalisations and/or indirect signs such as tracks, scats, diggings, or scratches (e.g., Friend 1978; Catling et al. 1997; Mills et al. 2002). Across all of these studies, the common findings are that different survey methods are useful for sampling particular fauna species, and no single approach accurately samples all species within a community. Therefore, as advocated by numerous previous researchers, surveys aimed at detecting multiple species must employ a suitable combination of survey methods (e.g., Laurance 1992; Brown and Nicholls 1993; Catling et al. 1997; Crosswhite et al. 1999; Lindenmayer et al. 1999; Ryan et al. 2002; Doan 2003). The selection of these methods should be influenced by the species or taxa of interest, but consideration in the survey design must also be given to dietary and habitat preferences, behavioural attributes, and body size of the target species (Mengak and Guynn 1987; Laurance 1992; Catling et al. 1997; Crosswhite et al. 1999; Lindenmayer et al. 1999; Mills et al. 2002).

In addition to differences in detection success, each fauna survey method varies in the required degree of effort (person hours) and the cost expended to detect target fauna. Consequently, the choice of survey method is commonly limited by a project's financial budget and time frame. It is important, therefore, that the method(s) selected will produce the greatest detection success within the time frame and whilst maintaining low overhead costs. Such information is particularly pertinent for surveys that aim to detect a range of species. Although a small number of authors have qualitatively discussed effort associated with various survey techniques (e.g., Catling et al. 1997; Crosswhite et al. 1999; Bisevac and Majer, 2002; Mills et al. 2002), we could find no study that explicitly examined and compared the detection success and quantitative costs associated with different survey methods for terrestrial reptiles and small mammals.

This paper compares six survey methods, for detecting terrestrial reptile and small mammal species, in terms of their relative detection success and costs of surveying. Three key questions were posed: (i) What is the relative success of each method for detecting reptiles and mammals? (ii) What is the cost associated with each method? (iii) What is the most cost efficient method or combination of methods? These questions are addressed using the wildlife survey results from a larger urban ecology research project conducted in Brisbane, Queensland. The main aim of this larger project was to determine the relative importance of habitat attributes at multiple spatial scales, for influencing the occurrence of native terrestrial reptiles and small mammals. As such, wildlife surveys were designed specifically to determine the occurrence of these target species within remnant forest fragments. The following analysis, therefore, is tangential to the main project aim, yet is important for expanding our understanding of effective and cost-efficient combinations of survey methods, and for providing a comparison of cost-effectiveness when selecting survey methods for future studies.

3.3. METHODS

3.3.1. Study Area and Survey Design

The study was conducted within the Brisbane City Council (BCC) local government area of south-east Queensland (153º2'S, 27ºE; area 1,220 km^2, population > 1 million) (Figure 3.1). Fifty-nine survey sites were established within lowland, remnant bushland fragments in the City's south and south-east suburbs (Figure 3.1). As a requirement of the larger research project, survey sites were located within Regional Ecosystem (RE) type 12.9-10.4, which is dominated by scribbly gum (*Eucalyptus racemosa*) woodland located on sedimentary rocks and sandy soils (Young and Dillewaard 1999).

As the main aim of the larger project was to investigate the influence of habitat attributes on species occurrence, sites were surveyed using the same trap layout. Each site measured 20 m x 45 m and was surveyed using three parallel transects 10 m apart, orientated perpendicular to the natural slope of the land. Each site was surveyed over three consecutive nights (a survey cycle) during fine weather in the spring/summer breeding season, using a combination of live-trapping and passive detection methods. Initial surveys of all 59 sites occurred in 2004, and repeat surveys of 51 sites were conducted in 2005. Eight sites were not repeat-surveyed due to recent fire or human interference. The order in which sites were repeat-surveyed was randomised in order to avoid surveying sites at the same time of season as the initial surveys, which may have biased detection success.

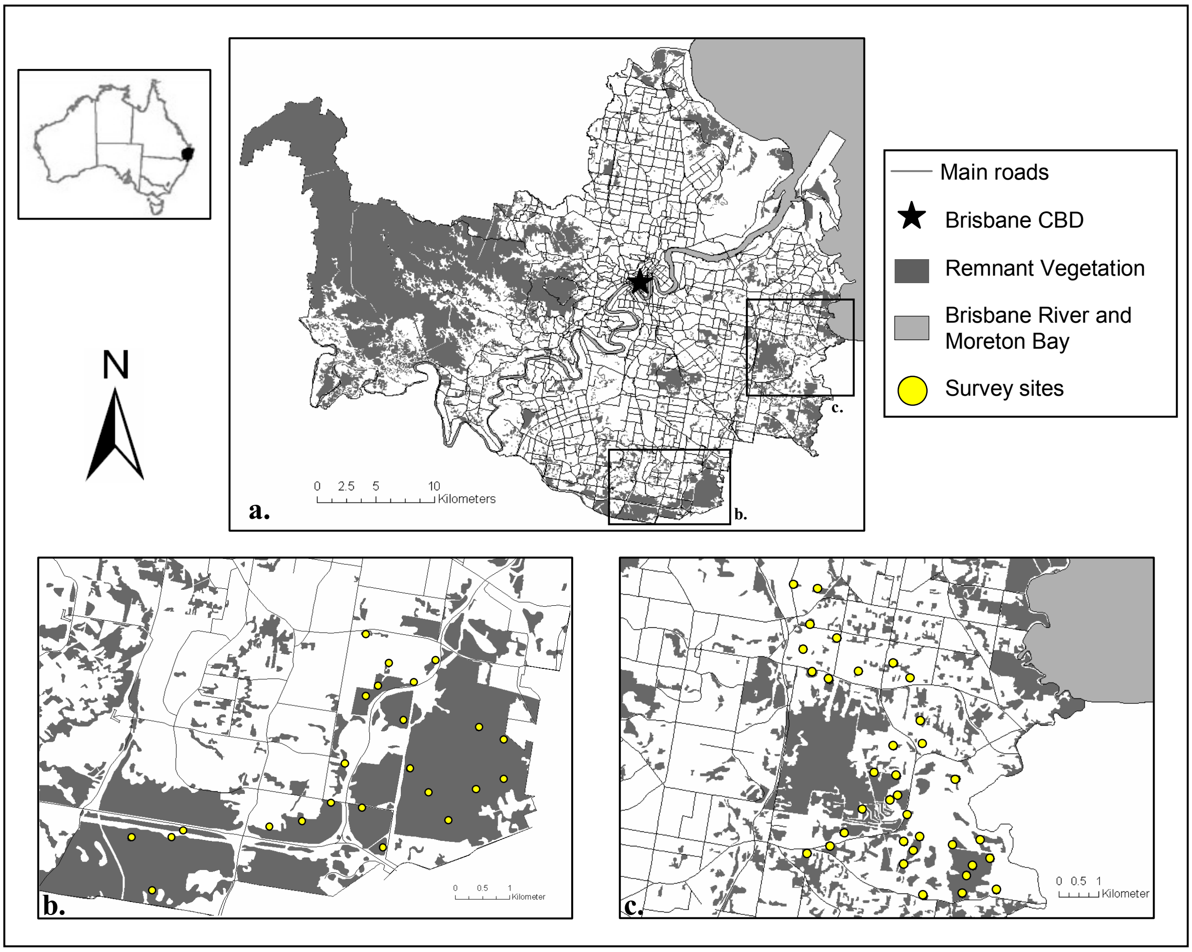


Figure 3.1. Map of study area showing the location of (a) Brisbane and the central business district (CBD), and site locations within (b) south suburbs, and (c) south-east suburbs. GIS data provided by Brisbane City Council. Remnant vegetation shown is derived from 1999 GIS data.

3.3.2. Trapping and Detection Methods

The selection of survey methods was based on initial pilot surveys, trap availability, and project budget and time constraints. Because of time constraints, we were not able to survey each site for more than three nights per survey. To compensate, we used more traps per site to increase trapping effort per night per site, thereby improving detection success within the short time frame. The complement of cage and Elliott traps available from The University of Queensland's Ecology Centre, were divided evenly between four sites, which were to be simultaneously surveyed during each survey cycle. A number of unsuitable cage traps were replaced with new traps, and pit-fall buckets, hair funnels, and associated preparatory equipment were then purchased within the constraints of the remaining project budget. Consequently, each site was surveyed using a combination of: eight wire cage traps, 10 Elliott traps, five dry pit-fall traps (10 L buckets), and three hair funnels.

Traps were spaced approximately 5 m apart along the three transects (Figure 3.2). Where possible, but without deviating more than 1 m from transects, traps were positioned alongside, on, or in logs, grass runways, or possible shelter sites in order to maximise the chances of being encountered by an animal (Sutherland 1996b; Cunningham et al. 2005). Cage traps, Elliott traps, and hair funnels were baited with the standard Australian mammal mixture of peanut butter, rolled oats, and honey, with vanilla essence also added (Menkhorst and Knight 2001). A piece of apple was also used as bait in the cage and Elliott traps. Pit-fall traps were not baited. Direct observations and scat collection occurred opportunistically during each site visit.

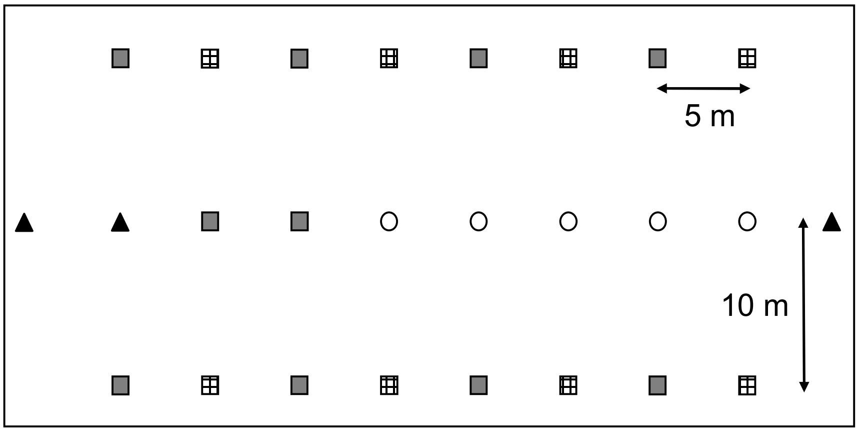


Figure 3.2. Schematic of trap layout along the three transects at each survey site. ▲ Hair funnels; ■ Elliott traps; ⊞ Cage traps; ○ Pit-fall traps.

Cage traps were used to detect medium-sized terrestrial mammals such as native rodents (e.g., *Rattus fuscipes* and *R. lutreolus*) and bandicoots (e.g., *Isoodon macrourus* and *Perameles nasuta*), whereas Elliott traps targeted small-bodied species such as dunnarts (*Sminthopsis spp.*) and antechinus (*Antechinus spp.*). The hair funnels (Faunatech Pty Ltd, Bairnsdale, Victoria) used to detect both small and medium-sized mammals differed in design from other hair sampling devices (e.g., Scotts and Craig 1988; Lindenmayer et al. 1999; Mills et al. 2002) in having only a single large opening that tapers to an enclosed bait chamber. A specialised wafer was attached to the upper inside surface of the funnel. The wafer was covered with a sticky substance ('faunagoo'), which replaces the double-sided tape used in previous similar traps to collect hair samples. Dry pit-fall traps were employed primarily to detect small reptile species, with direct observations being used to detect large-bodied reptiles that were unlikely to be detected by other methods. Scats were also collected for *ex situ* analysis.

Cage and Elliott traps were set and baited each afternoon before sunset, checked for captures before dawn the following morning, and then closed during the day. Captured animals were identified to species-level using a field guide (Menkhorst and Knight 2001), photographed, weighed, sexed, and immediately released at the point of capture.

Hair funnels were set and baited at the start of the three day survey cycle and left undisturbed until collection at the end of the survey cycle. All wafers with hair samples were sent for identification by one of two independent experts (initial surveys: Michiala Bowen; repeat surveys: Barbara Triggs). Hair sample were identified to genus or species level, and were categorised as either “definite” or “probable”. Only definite species identifications were used for subsequent data analyses.

Dry pit-fall traps (i.e. no chemicals/preservatives used) were established at least one week prior to site surveys to allow species and habitats to recover from the localised disturbance that occurred as a result of digging holes for pit-fall traps. Given the main aim of the larger project, care was taken to minimise habitat disturbance. For this reason, we did not use drift fences, so as to avoid disturbance to dense ground vegetation and fallen woody debris at many of the survey sites. This differs from previous studies which have used drift-fences in conjunction with pit-fall traps to improve capture success (e.g., Mengak and Guynn 1987; Friend et al. 1989; Crosswhite et al. 1999; Menkhorst and Knight 2001). Pit-fall traps were open for the duration of the survey cycle and were checked for captures each morning and afternoon. Captured animals were identified to species-level using a field guide (Wilson 2005), photographed, weighed, sexed (if possible), and immediately released at the point of capture. Between the initial and repeat wildlife survey periods, lids were securely fitted to each pit-fall trap to prevent captures. Direct observations and scat collection occurred opportunistically throughout all site visits to identify target species. Where possible, and if necessary, species were caught (by hand) to ensure accurate identification. All scats collected were identified by the same experts who analysed the hair samples. Only 'definite' species identifications from scat analysis were used in subsequent analyses.

The relative success of each survey method was determined by evaluating the total number of species detected by each method across all sites. The total number of "unique" species detected by each method was also examined. Unique species were those species detected by only one survey method (*sensu* Doan 2003). Species detections from the initial and repeat surveys were collated for analyses.

3.3.3. Cost Analysis

The cost of each survey method was calculated independently based on the cost required to survey all sites over the total survey period (2004 and 2005). Four main areas of cost expenditure were considered for each method: equipment, bait/analysis, personnel, and travel.

Equipment costs included expenditures for acquiring traps and the additional items required to prepare each trap. Because some traps were reused for surveys, a 20% depreciation rate based on the 2004 purchase price for each trap type was used to represent equipment costs. Additional preparatory expenses included shade cloth for cage traps, non-reusable hair wafers for hair funnels, and the hire of a motorised auger for digging holes for pit-fall traps. As shade cloth was replaced at the start of the initial and repeat survey periods, this cost was calculated for each survey period using the 2004 retail price from a major discount hardware store. A total of three hair wafers were used to survey each site for each survey period (a total of 330 wafers), and as wafers are non-reusable across different survey sites, the 2004 purchase price (Faunatech Pty Ltd, Bairnsdale, Victoria) was used to calculate this cost. The cost of hiring a motorised auger for 12 days (5 sites established each day) was calculated on 2004 daily hire rates from a major equipment hire business in Brisbane.

Bait/analysis costs covered bait expenses as well as hair and scat analysis charges. The cost of bait was calculated based on retail costs (from a major supermarket) for ingredients required to bait traps over the entire survey period. Cage traps and hair funnels used larger peanut butter balls than Elliott traps and the price was adjusted accordingly. For each survey night, fresh bait was used in cage and Elliott traps, whereas the hair funnel bait was left unchanged during the survey cycle. The charge cost for expert analyses of hair and scat samples differed between the two experts and so an average charge was used to calculate costs of hair and scat analyses.

Personnel costs were calculated for two people based on The University of Queensland's minimum hourly wage for a casual research assistant of $19 per hour plus 15.5% on-costs (all prices are given in Australian dollars). Two people were included in personnel calculations as this is the minimum personnel required to meet The University of Queensland's field work safety standards. Personnel cost calculations incorporated the time taken to prepare traps, establish traps at a site, set and check traps each survey day, and remove traps at the end of the survey cycle. Although observation and scat collection were not standardised, an estimated 30 minutes per site visit was calculated for each person.

Travel costs were based on The Ecology Centre's (The University of Queensland) vehicle hire charge of $0.50/km. A total of six return trips, each averaging 30 km, were required each survey cycle to set, check, and remove traps, or conduct direct observations/scat collection. With a maximum of four sites being surveyed simultaneously in each survey cycle, 28 return trips were required to survey all sites for both survey periods. The number of return trips was calculated independently for each survey method. Therefore, cage trapping, Elliott trapping, direct observation, and scat detection, each required six return trips during each survey cycle, and 168 return trips over the two survey periods. An extra 12 return trips for digging pit-fall trap holes were included in travel costs associated with pit-fall trapping.

3.3.4. Cost Versus Success

The average number of species and unique species detected per dollar was calculated for each survey method based on the average number of species and unique species detected by each survey method per site, and the average cost per site for each method. Comparing these results enabled us to identify the most effective and efficient method, or combination of methods, for surveying terrestrial reptiles and small mammals in the Brisbane case study.

3.4. RESULTS

3.4.1. Species Detected

A total of 19 target reptile species (eight families), and nine target mammal species (three families), were detected during the overall survey period (Table 3.1). A number of non-target species were also detected, including large, terrestrial native mammals (wallabies and kangaroos), invertebrates, amphibians, birds, arboreal marsupials, and exotic mammals and reptiles.

3.4.2. Survey Method Success

Each survey method successfully identified at least two species, with certain methods detecting up to 14 species (Figure 3.3a). As expected, each method was mainly suited to detecting either mammals or reptiles (Figure 3.3a). Although there was a degree of overlap in the species detected by each survey method, all methods detected at least one species not detected by another method (Figure 3.3b).

Table 3.1. Collated species list showing the method/s by which each species was detected. Numbers indicate the total number of individuals captured, or, for hair funnels, the total number of wafers containing hair samples. Note that marking was not used so individual captures are not necessarily different animals. CT = cage trap; ET = Elliott trap; PF = pit-fall trap; HF = hair funnel; Obs. = direct observation; S = scat detection/analysis.

				Survey Method					
	Family Group	**Scientific Name**	**Common Name**	**CT**	**ET**	**PF**	**HF**	**Obs.**	**S**
Native Mammals	Dasyuridae	*Antechinus flavipes*	Yellow-footed antechinus		7				
		Antechinus subtropicus	Subtropical antechinus				2		
		Planigale maculata	Common planigale			2			
		Sminthopsis murina	Common dunnart		12				
	Muridae	*Melomys* sp.	Likely: Grassland melomys						1
		Rattus fuscipes	Bush rat	2					
		Rattus lutreolus	Swamp rat	1					
	Peramelidae	*Isoodon macrourus*	Northern brown bandicoot	15			3		1
		Perameles nasuta	Long-nosed bandicoot				2		
Native Reptiles	Agamidae	*Diporiphora australis*	Tommy round-head			3		2	
		Physignathus lesuerii	Eastern water dragon					1	
		Pogona barbata	Bearded dragon			1		4	
	Colubridae	*Dendrelaphis punctulata*	Common tree snake					1	
	Elapidae	*Pseudechis porphyriacus*	Red-bellied black snake					1	
	Gekkonidae	*Diplodactylus vittatus*	Eastern stone gecko			6			
	Pygopodidae	*Lialis burtonis*	Burton's snake-lizard					1	
	Pythonidae	*Morelia spilota*	Carpet python	1					
	Scincidae	*Anamalopus verreauxii*	Verreaux's skink			2			
		Calyptotis scutirostrum	Scute-snouted calyptotis skink			27			
		Carlia foliorum	Tree-base litter-skink			3		2	
		Carlia pectoralis	Open-litter rainbow skink			1		4	
		Carlia vivax	Storr's rainbow skink			7		6	
		Cryptoblepharus virgatus	Fence skink			14		5	
		Ctenotus taeniolatus	Copper-tailed skink			2			
		Eulamprus quoyii	Eastern water skink					1	
		Lampropholis amicula	Secretive skink			2		1	
		Lampropholis delicata	Garden skink			54	1	4	
	Varanidae	*Varanus varius*	Lace monitor					5	

(a)

(b)

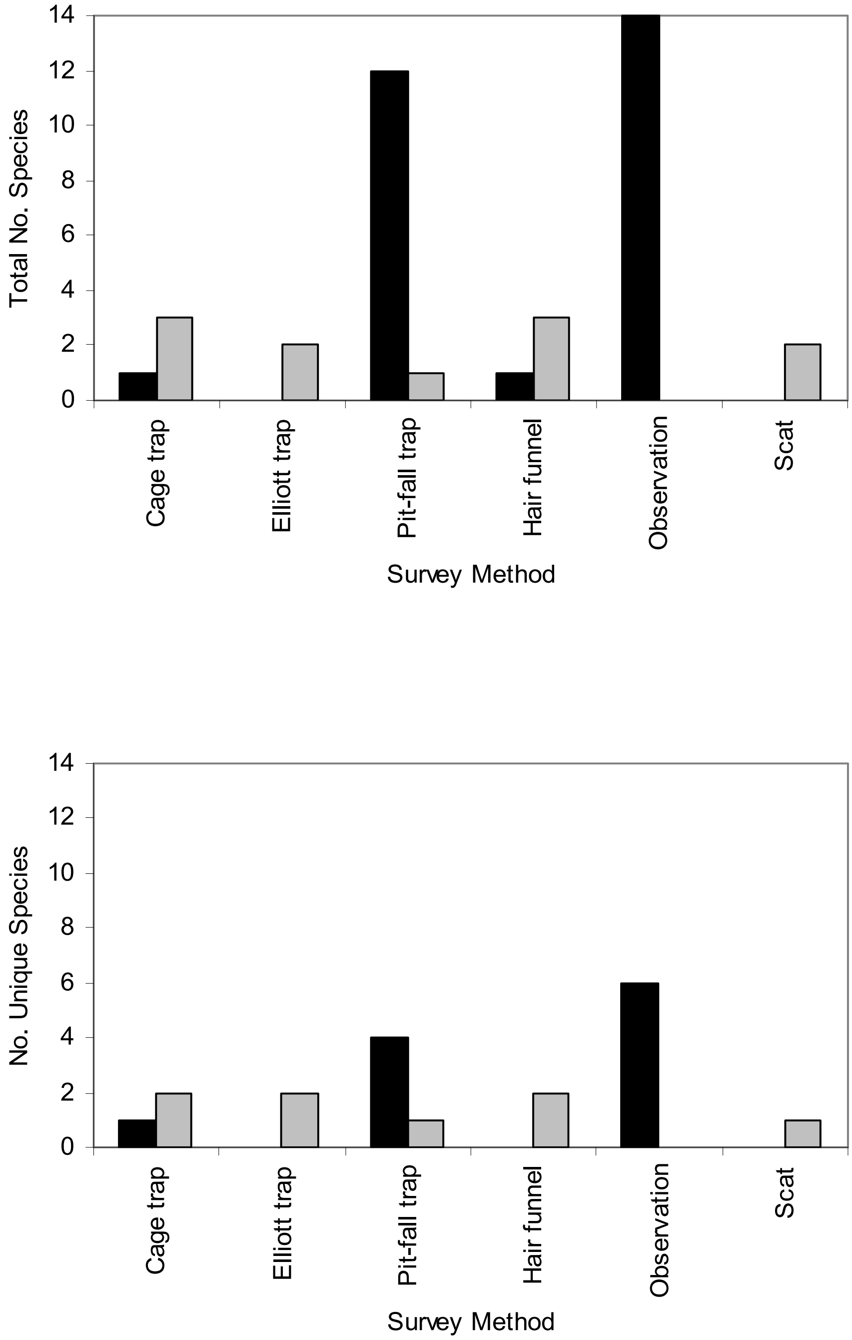


Figure 3.3. Survey method detection success. (a) Total number of species detected by each survey method, and (b) total number of unique species detected by each survey method. ■ Reptiles; □ Mammals.

Cage traps

2568 potential captures (i.e. 321 trap nights x 8 cage traps) produced 278 successful captures (capture success = 10.8%). Target reptile and mammal species comprised 6.8% (*n* = 19) of the total cage trap captures. Collectively, these species represented three family groups and four species (Table 3.1). With the exception of one reptile capture (carpet python, *Morelia spilota*) all target species captured by this method were medium-sized mammals. The northern brown bandicoot (*Isoodon macrourus*) was the most commonly captured target mammal (*n* = 14 across 7 sites). Bush rats (*Rattus fuscipes*) were captured on two nights at a single site, and the swamp rat (*Rattus lutreolus*) and carpet python were identified at one site each, from a single capture. The bush rat, swamp rat, and carpet python were not detected by any other survey method (Table 3.1).

Captures of non-target species included: the common brushtail possum (*Trichosurus vulpecula*) (*n* = 124), exotic black rat (*Rattus rattus*) (*n* = 100), introduced cane toad (*Bufo marinus*) (*n* = 26), Australian magpie (*Gymnorhina tibicen*) (*n* = 6), grey butcherbird (*Cracticus torquatus*) (*n* = 2), and Torresian crow (*Corvus orru*) (*n* = 1). Overall, cage traps were found closed without captures 7.2% of the time (*n* = 184).

Elliott traps

A total of 3210 potential Elliott trap captures produced 55 captures for an overall trap capture success of 1.7%. Of these captures, 34.5% (*n* = 19) were target species that represented two mammal species from the Dasyuridae family (Table 3.1). Common dunnarts (*Sminthopsis murina*) were captured more frequently and across a greater number of sites (*n* = 12 across seven sites) than yellow-footed antechinus (*Antechinus flavipes*) (n = 7 across four sites). Both species were detected only by this method (Table 3.1). Non-target species captured by Elliott traps included: cane toads (n = 16), house mouse (*Mus musculus*) (n = 14), giant white-kneed king cricket (*Australostoma australasiae*) (n = 4), centipede (n = 1), and a juvenile common brushtail possum (n = 1). Elliott traps were found closed without captures 4.1% of the time (n = 133).

Pit-fall traps

The 550 pit-fall traps used during the survey period were left open throughout survey cycles, and were checked twice daily for captures (i.e. five checks per bucket, per survey cycle), giving a total of 2750 possible identification opportunities over the total survey period. Of these opportunities, 142 or 5.1 % were successful for identifying vertebrate species. Of these, 12 were target reptile species representing three families, and one was a target mammal species (Table 3.1). The reptile species captured were mostly small-bodied skinks and lizards, but also included one juvenile of a larger-bodied agamid (bearded dragon, *Pogona barbata*). Four reptile species were not detected by any other method: Verreaux's skink (*Anamalopus verreauxii*), scute-snouted calyptotis skink (*Calyptotis scutirostrum*), copper-tailed skink (*Ctenotus taeniolatus*), and eastern stone gecko (*Diplodactylus vittatus*) (Table 3.1). The only mammal species captured in pit-fall traps was the common planigale (*Planigale maculata*), which was identified from two captures of individuals at two different sites. The common planigale was detected during surveys by this method only (Table 3.1).

Non-target species captures in pit-fall traps were commonly invertebrate species such as ants, spiders, snails, and crickets. Non-target vertebrate by-captures were all amphibians: cane toads (n = 16), ornate burrowing frogs (*Limnodynastes ornatus*) (n = 7), and copper-backed broodfrogs (*Pseudophryne raveni*) (n = 5).

Hair funnels

Over the total 321 trap nights, a total of 330 wafers were used (three for each site survey). Of these 330 wafers, 84 (25.5%) returned hair samples, 17 (20.2%) of which contained hair that was identifiable to the species level, with seven of these representing target species. Three target mammal species were positively identified from hair samples: subtropical antechinus (*Antechinus subtropicus*), northern brown bandicoot (*Isoodon macrourus*), and long-nosed bandicoot (*Perameles nasuta*). The subtropical antechinus and long-nosed bandicoot were identified only by this method (Table 3.1). Although hair funnels specifically target mammal species, a live reptile by-catch of a single garden skink (*Lampropholis delicata*) was found on one wafer at one site.

Direct observations and scat analyses

Opportunistic species identifications through direct observations, and scat collection and analyses, respectively identified a total of 14 target reptile species and two target mammal species (Table 3.1). Direct observations identified six reptile species not detected by any other method (Table 3.1). Fifty scat samples were collected, 60% (n = 30) of which were identified as ‘definite’ species, but only one was a target species (Table 3.1). The other target mammal identified from scat samples was a *Melomys spp.*, and was only able to be ‘definitely’ identified to the genus level (Table 3.1), although the grassland melomys (*M burtoni*) was considered 'probable'. As this was the only detection of a representative species from this genus, the genus record was included in subsequent analyses.

3.4.3. Cost of Detection Methods

There were substantial variations in cost among the survey methods. Total costs to survey a site over the entire survey period ranged from \$4 807 for hair funnels to \$12 284 for cage traps (Figure 3.4a). The most expensive methods after cage trapping were Elliott trapping (\$11 953), followed by pit-fall trapping (\$10 067), scat detection and analysis (\$7 703), and direct observation (\$7 453) (Figure 3.4a). The largest proportions of cost for each method were attributed to travel and personnel expenses (Figure 3.4b). Travel expenses comprised roughly 60% of the total costs for scat detection and direct observations (65.4% and 67.6%, respectively), 53.6% of the total cost for pit-fall trapping, approximately 40% for cage and Elliott trapping (41% and 42.2%, respectively), and 35% for hair funnels (Figure 3.4b). Personnel expenses comprised just over 50% of the total costs for cage and Elliott trapping (52.8% and 53.8%, respectively), 41.8% of the costs for hair funnels, and between 30-40% for pit-fall trapping, direct observations, and scat detection (37.5%, 32.4%, and 31.3%, respectively) (Figure 3.4b). Equipment costs incurred the highest proportion of total costs for hair funnels (14.9% of total cost) due to new wafers being required for each site survey (Figure 3.4b). Operating costs also represented the highest proportion of overall costs for hair funnels, due to the combination of bait and hair analysis expenses (Figure 3.4b). Direct observations and scat detection incurred no equipment costs, and no operating costs were involved with pit-fall trapping and direct observations.

(a)

(b)

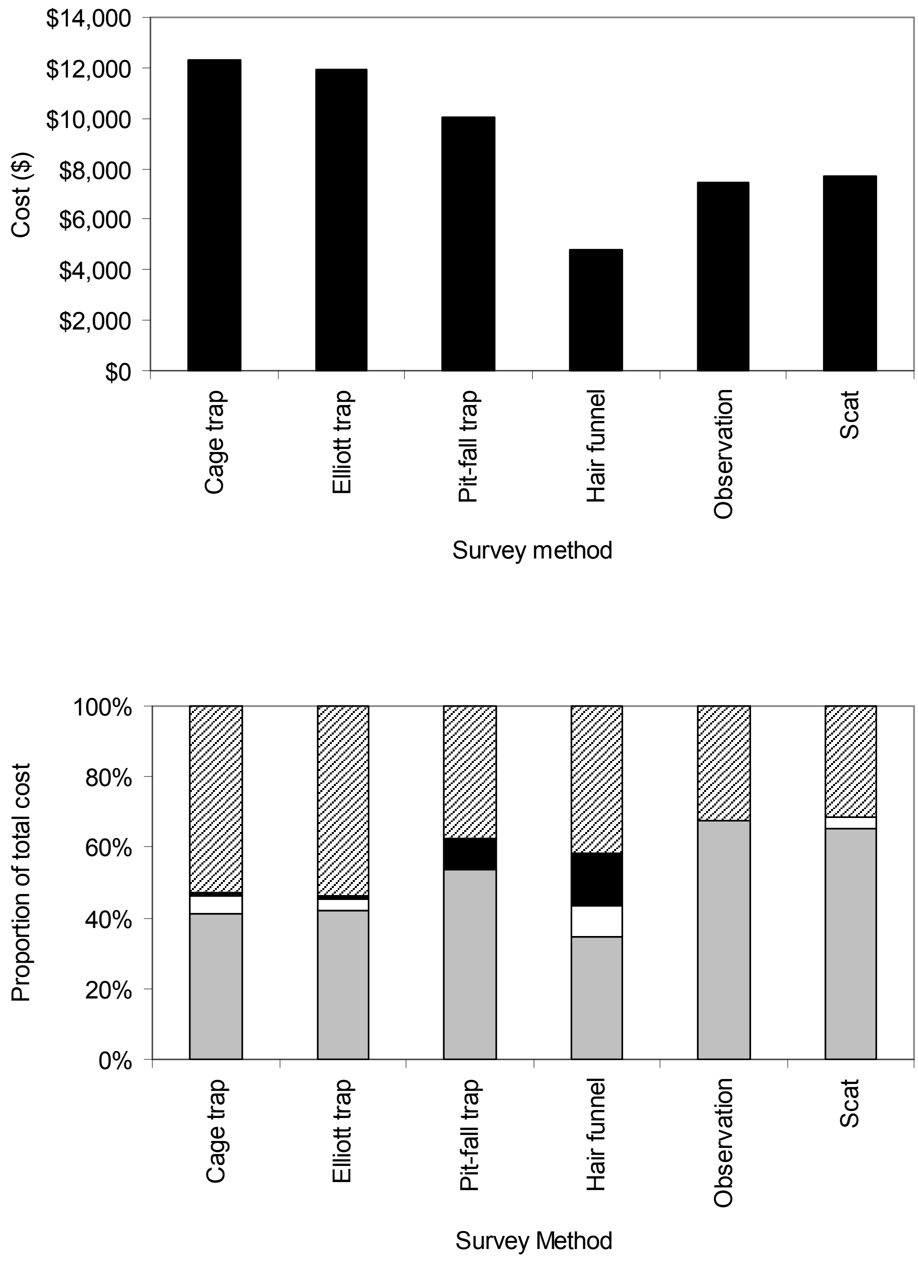


Figure 3.4. Relative cost of each survey method in isolation as used during the Brisbane case study. (a) Total cost for each method to survey all sites over the entire survey period; (b) Proportion of total cost for each method for expenses relating to: ▨ personnel, ■ equipment, □ bait/analysis, and ■ travel. Equipment costs are calculated at a 20% depreciation rate of the 2004 purchase price.

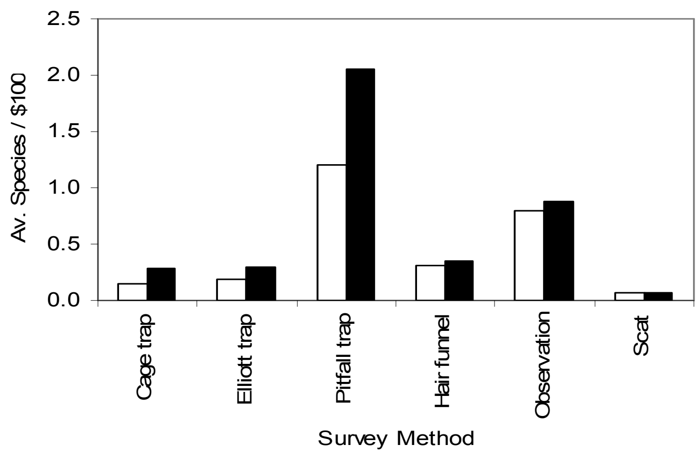
3.4.4. Effectiveness and Cost Efficiency

Overall, pit-fall trapping was the most effective and cost efficient survey method, detecting both the highest number of species (n = 0.012), and captures (n = 0.0206), per dollar (Figure 3.5a). Direct observations, followed by hair funnels were the next most successful and efficient survey methods (Figure 3.5a). Elliott traps and cage traps were roughly similar in their cost per species and cost per capture, and although their average capture success per site were both similar to hair funnels, they both detected fewer species per dollar than did hair funnels (Figure 3.5a). Comparatively, scat detection/analysis had the lowest detection success for money spent, with 0.0007 detections per dollar (Figure 3.5a).

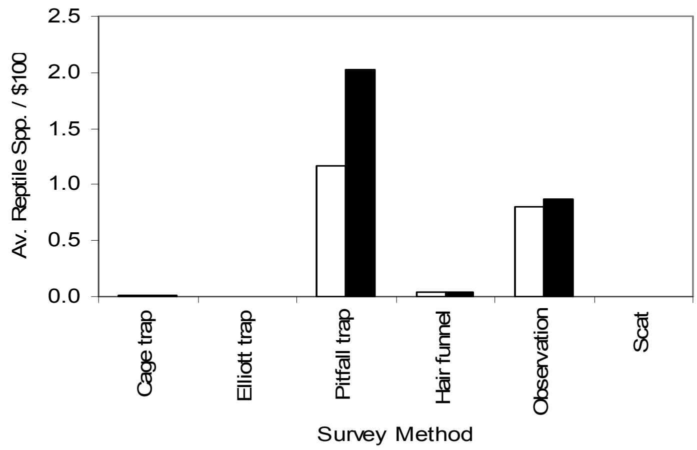
For reptiles, the most successful survey methods mirrored those for overall detection costs, with pit-fall traps followed by direct observations producing the highest species detection (n = 0.0117 and 0.008, respectively) and capture success (n = 0.0202 and 0.0088, respectively) per dollar (Figure 3.5b). Direct observation per dollar values for the number of species detected and the number of individuals captured were equal to those shown in Figure 3.5a, as mammals were not detected using this method (Figure 3.3a). The remaining four methods failed to detect reptiles, with the exception of a single by-catch in one hair funnel and one cage trap.

For mammals, hair funnels, Elliott trapping, and cage trapping all returned roughly equivalent individual captures per dollar (n = 0.0031, 0.003, 0.0027, respectively), but there were substantial differences between the number of species detected by each method (Figure 3.5c). Hair funnels detected the most number of species per dollar (n = 0.0027), followed by Elliott trapping (n = 0.0019), and then cage trapping (n = 0.0014) (Figure 3.5c). This implies that hair funnels incurred the lowest cost per species detected and per capture. In comparison, scat

(a)



(b)



(c)

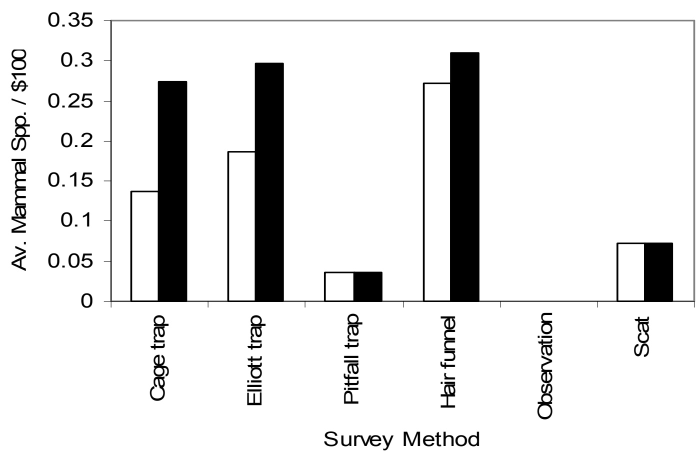


Figure 3.5. Optimal survey methods, calculated as the average species detection and capture success of each survey method per site, per dollar. For clarity, the cost of success is shown as success per hundred dollars for: (a) all target species; (b) reptile species; and (c) mammal species. For all graphs: □ average number of species detected; ■ average number of captures.

detection/analysis and pit-fall trapping incurred higher costs per species detection and individual capture than the other methods, with the exception of direct observations which were not useful for detecting any target mammal species.

3.5. DISCUSSION

The selection of fauna survey methods will vary depending on the species/taxa of interest, the habitat/topography of survey sites, and the financial, resource, and time constraints of the project. Consequently, it is difficult to recommend a particular method or combination of methods that will be suitable for all survey projects. However, given the thousands of dollars spent on terrestrial surveys each year, it is useful to know the costs and detection successes/failures of previous studies in order to facilitate informed method selection in future surveys.

The following section discusses the success and cost efficiency of the survey methods used during the Brisbane case study survey of native terrestrial reptiles and small mammals. Our findings are not presented as a definitive, ‘best choice’ of survey methods for all future surveys, but rather, as a comparison for improving method selection in future surveys. We recommend that all researchers should report on the successes and costs of independent fauna surveys in order to broaden our understanding of useful and cost efficient trapping combinations for different species and within various habitat types. Doing so will be important for refining the survey method selection process.

No one survey method independently detected all species recorded during the surveys, and each method was important for detecting between 1 – 6 species that were not detected by any other method. There were distinct biases between methods in terms of the fauna groups (reptiles vs. mammals) most successfully detected. Pit-fall traps and direct observations were more successful in detecting reptile species, whereas hair funnels, cage traps, Elliott traps, and scat detection/analysis, were more useful for detecting mammal species. Such differences in effectiveness correspond with the purpose for which each method was designed (Sutherland 1996a), and further supports the conclusion by Laurence (1992), that the use of only a single method in fauna community surveys will "…be biased toward a non-random subset of species in the community..." (p. 654). Therefore, when surveying fauna communities, we believe it is essential that a combination of survey methods is employed in order to adequately sample a range of species. This finding is also consistent with previous studies that have advocated the use of a combination of techniques for surveying a range of wildlife species (e.g., Mengak and Guynn 1987; Catling et al. 1997; Lindenmayer et al. 1999). Given the range of possible survey methods, coupled with commonly limiting survey resources, it is useful to know not only which method combinations have been successful in previous studies, but also, which were the most cost effective.

We found a considerable difference among survey methods in terms of the cost required for detecting ground-dwelling mammal and reptile species. However, these calculations were based on the complement of traps used during the Brisbane case study and, therefore, costs are likely to vary in other studies depending on survey design (i.e. the number of traps and sites, duration of active searches, and number of days surveyed), personnel costs, distance to survey sites and associated transport costs, and varying costs for consumables (e.g., bait ingredients, equipment purchase/hire). To help provide a direct comparison between the success and cost of each survey method, success was interpreted in terms of 'value for money' (i.e., the average number of species detected and individual captures per dollar for each survey method). We found that the methods that were most successful for detecting reptiles and mammals, were often those that detected the most mammal and reptile species and individual captures at the least cost.

3.5.1. Reptiles

Based on the results of this study, the most effective trapping methods for terrestrial reptile surveys are a combination of pit-fall traps and direct observations. Despite being opportunistic rather than standardised, direct observations detected more overall reptile species, as well as more unique species than pit-fall trapping. However, pit-fall trapping produced the highest overall species detection and capture success for the least amount of money. The unique species detected by each of these methods was biased by body size and behaviour. Large-bodied reptiles such as lace monitors, snakes, and dragons that were unlikely to be captured in pit-fall traps were more frequently detected by direct observation.

In comparison, the species detected by pit-fall traps were small-bodied, cryptic, nocturnal, and/or fossorial reptiles such as small skinks, eastern stone geckos, and Verreaux's skinks. Further, the relative species detection success of pit-fall traps is expected to have been even greater than direct observations had it been possible to use drift fences and larger traps (buckets). Crosswhite et al. (1999), for instance, showed that more reptiles were captured using pit-fall traps combined with drift-fences than through active searches, and Catling et al.

(1997) concluded that increasing the size of the buckets used as pit-fall traps would likely have increased the capture success of larger-bodied species that were able to otherwise escape. However, increasing the trap size and/or incorporating drift fences would also have increased the costs associated with pit-fall trapping.

Similarly, given that all direct observations were opportunistic, it may be expected that a dedicated, time-constrained active search period would have identified additional species. However, in accordance with reports by Crosswhite et al. (1999) and Ryan et al. (2002), pit-fall trapping is still likely to have detected the highest number of overall species and captures. Furthermore, additional costs and limitations associated with personnel time and experience is likely to influence the accuracy of active search results, thereby potentially offsetting any increase in species detections through time-constrained searches (Crosswhite et al. 1999; Silveira et al. 2003).

Based on the cost efficiency and detection success of pit-fall trapping and direct observations, pit-fall trapping was the best technique in the Brisbane study for surveying small-bodied terrestrial reptiles. For future studies, where habitats and budgets allow, drift fences should also be used to increase capture success (Friend et al. 1989; Moseby and Read 2001). The larger-bodied reptiles, however, were best detected by direct observations. Therefore, for surveying terrestrial reptile communities, as was the aim in the Brisbane study, pit-fall traps and direct observations were the best combination of survey methods for detecting the highest number of total species, per dollar. Furthermore, when used in combination, the travel costs were approximately 50% less than that calculated independently for each survey method. This may compensate for increased expenses associated with the inclusion of drift-fences and standardised search times.

3.5.2. Mammals

Mammal species were detected by five of the six survey methods. Optimum trapping methods for terrestrial, small- and medium-sized mammal surveys were hair funnels, followed by Elliott trapping, cage trapping, and then scat analysis and pit-fall trapping. Although hair funnels, Elliott traps, and cage traps all detected the same number of unique species, the cost effectiveness of these captures varied considerably. Hair funnels were the most economical method for targeting mammals, detecting more species and captures per dollar than the other survey methods. The relative cost effectiveness of hair funnels is most likely related to the advantages this method has over live trapping methods in being less labour-intensive, able to detect more than one species or individual per funnel, and able to capture hair from species of various body-sizes (Lindenmayer et al. 1999; Mills et al. 2002). However, unlike live trapping methods, differentiating between individuals of the same species is not possible using hair funnels, thereby limiting the usefulness of this method for surveys aimed at determining species abundance. Furthermore, accurately identifying hair samples to the species level is a difficult process that is susceptible to hair samples either being incorrectly identified or unable to be identified. Lobert et al. (2001), for instance, tested the accuracy of hair identification by two independent experts and found that almost half of the samples analysed involved some degree of identification error. Furthermore, the identification of dog hair from a hair funnel used in Lindenmayer et al.'s (1999) study was questionable given the hair sample was from an arboreal funnel (see also: Kavanagh and Stanton 1998). However, the definite species identifications from hair samples collected during the Brisbane study were considered reasonable based on species habitat requirements and range, as well as the extensive experience of both experts. The only species identification that was slightly questionable was the subtropical antechinus, which is generally considered to be a rainforest species. However, the hair samples were reanalysed several times and the expert was confident with the identification each time.

There were clear associations between the body size of mammals and the success of the methods for detecting different species. Elliott traps, given their size, detected only small-bodied mammal species, with the exception of a single by-catch of a juvenile brushtail possum at one site. Similarly, small-bodied species, such as dunnarts and antechinus were not captured in cage traps, probably due to the lower sensitivity of cage trap treddles and the light weight of small-bodied species. Cage traps were, however, successful in capturing medium-sized species such as rodents and bandicoots, whose body size prevented captures in Elliott traps. Due to their tapered design and large surface area for capturing hair samples, hair funnels detected both medium- and small-bodied terrestrial mammals, including species not captured by live trapping methods (e.g., long-nosed bandicoot and subtropical antechinus).

Furthermore, although more commonly used for reptile surveys, pit-fall trapping was the only method to detect the occurrence of planigales during this study. This is consistent with previous studies that have found pit-fall traps to be useful for detecting small, elusive mammal species that rarely enter other traps (e.g., Milledge 1991; Laurance 1992; Catling et al. 1997; Menkhorst and Knight 2001; Clemann et al. 2005). The use of deeper pit-fall buckets in conjunction with drift fences may have increased the number of small mammals detected by this method. This may, however, have influenced the identification of reptile species captured in pit-fall buckets, if captured reptiles were predated by small carnivorous mammals. Consequently, although only successful for detecting planigales in the Brisbane case study, pit-fall traps may be useful in other surveys that are targeting small, elusive mammals, but may be somewhat counter-productive if reptiles are also being targeted.

Scat detection and analysis also identified one small mammal (*Melomys spp.*) not detected by any other method. However, these scat samples were not able to be positively identified to the species level, suffering the same analysis difficulties as hair sample identifications. Based on our results, both pit-fall trapping and scat detection/analysis methods would require a substantial cost outlay in order to detect as many species and individual captures as the other methods. As such, these methods are considered most useful as complementary, rather than primary, survey methods. If they had not been included, two species would not have been detected. However, the funding allocated to these methods could have been used to increase the survey-intensity of more optimal survey methods, or exploring the use of additional bait types, such as meat products, to help increase the range of species detected (e.g., herbivores, omnivores, and carnivores) (Laurance 1992; Mills et al. 2002). Such trade-offs are important considerations for future studies.

3.5.3. Conclusion

Although tangential to the primary aim of the larger research project, the information presented in this paper about trapping success and cost efficiency is important for broadening our understanding of effective and efficient survey methods. We urge other researchers to also report their survey design, cost output, and detection successes and failures. Further, surveys designed to explicitly determine optimal combinations of survey methods for different species and within various habitat types will be a valuable contribution to the current knowledge base.

(a) Grassland melomys (*Melomys burtoni*)

- Family: Muridae
- Selective omnivore, nocturnal rodent.
- Size range: HBL 90-160 mm, TL 90-175 mm, 26-124 g
- Distribution: Queensland, New South Wales, Northern Territory, Western Australia.
- In Brisbane: rarely encountered, restricted to suitable remnant habitat in peri-urban landscapes.

Photo from: http://www.griffith.edu.au/centre/cics/content/members/Castley.htm

(b) Tommy round-head (*Diporiphora australis*)

- Family: Agamidae
- Insectivorous, diurnal dragon.
- Average size: SVL 50 mm.
- Distribution: Queensland.
- In Brisbane: restricted to suitable remnant bushland habitat in peri-urban landscapes.

In this photo a Tommy round-head is basking on a fire damaged log.

(c) Eastern water dragon (*Physignathus lesuerii*)

- Family: Agamidae
- Omnivorous, diurnal dragon.
- Average size: SVL 245 mm.
- Distribution: Queensland, New South Wales, Victoria, South Australia.
- In Brisbane: abundant along waterways, including in inner suburbs.

Photo from: http://www.benel.com/powershot/potw2.php?weeknum=200318

(d) Eastern bearded dragon (*Pogona barbata*)

- Family: Agamidae
- Omnivorous, diurnal dragon.
- Average size: 250 mm
- Distribution: Queensland, New South Wales, Victoria, South Australia
- In Brisbane: abundant in bushland and urban gardens, including in inner suburbs

In this photo, a bearded dragon is basking on a large log in remnant habitat.

Plate 4. Melomys and dragons: (a) Grassland melomys; (b) Tommy round-head; (c) Eastern water dragon; (d) Bearded dragon.

Chapter 4

HABITAT STRUCTURE IS MORE IMPORTANT THAN VEGETATION COMPOSITION FOR LOCAL-LEVEL MANAGEMENT OF NATIVE TERRESTRIAL REPTILE AND SMALL MAMMAL SPECIES LIVING IN URBAN REMNANTS: A CASE STUDY FROM BRISBANE, AUSTRALIA

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4.1. ABSTRACT

As urban areas continue to expand and replace natural and agricultural landscapes, the ability to manage and conserve native wildlife within urban environments is becoming increasingly important. To do so we first need to understand species responses to local-level habitat attributes in order to inform the decision-making process and on-ground conservation actions. Patterns in the occurrence of native terrestrial reptile and small mammal species in 59 sites located in remnant urban habitat fragments of Brisbane City were assessed against local-level environmental characteristics of each site. Cluster analysis, multi-dimensional scaling ordination, and principal axis correlation were used to investigate relationships between species occurrences and environmental characteristics. Native reptiles were most strongly associated with the presence of termite mounds, a high amount of fallen woody material and a moderate amount of weed cover. Native small mammals were most strongly associated with the presence of grass trees (*Xanthorrhoea spp.*), and both reptiles and small mammals were negatively influenced by increased soil compaction. Significant floristic characteristics were considered to be important as structural, rather than compositional, habitat elements. Therefore, habitat structure, rather than vegetation composition, appears to be most important for determining native, terrestrial reptile and small mammal species assemblages in urban forest fragments. We discuss the management implications in relation to human disturbances and local-level management of urban remnants.

Key Words: Grass trees, fallen wood, termite mounds, soil compaction, weed cover, ordination, management.

4.2. INTRODUCTION

Over the past 50 years, habitat loss and fragmentation have had the greatest influence on terrestrial ecosystems and biodiversity worldwide (Sala et al. 2000; Millennium Ecosystem Assessment 2005). Agricultural land use has previously been regarded as the primary cause of habitat loss and fragmentation, yet as areas of suitable arable land are exhausted, agricultural expansion is declining (Millennium Ecosystem Assessment 2005). Comparatively, urban development is expanding rapidly, encroaching on agricultural and natural, non-arable, landscapes (Levia and Page 2000; Lugo 2002). It therefore seems likely that urban development will surpass agricultural land use as the primary anthropogenic driver of land use change, and habitat loss and fragmentation.

Areas deemed suitable for urban development often coincide with those areas that also support high native species richness and endemism (Lugo 2002). The impacts of urbanisation on fauna populations are multi-causal and multi-scaled, altering the *in situ* structure and composition of habitat fragments, as well as their spatial patterning within the landscape context. These habitat modifications potentially have important consequences for concomitant fauna assemblages, with significant differences being apparent between urban and pre-urban assemblages (Jones and Wieneke 2000, van der Ree 2004, Tait et al. 2005). Within the urban matrix, introduced species and a handful of native generalist species tend to dominate, whilst habitat and dietary specialists and migratory species tend to decline in numbers or become locally extinct (How and Dell 1993, 2000; White and Burgin 2004; Tait et al. 2005).

This problem is particularly pertinent to Australia, where more than 92% of the human population currently resides in urban areas (United Nations 2006). For native fauna assemblages, native terrestrial reptile and small mammal species are considered to be the fauna groups most sensitive to urbanisation and its associated disturbances (How and Dell 1993, 1994, 2000; van der Ree 2004; White and Burgin 2004; Tait et al. 2005). Although many native mammals and reptiles are negatively influenced by urban development, some native mammal and reptile species, such as the common brushtail possum (*Trichosurus* *vulpecula*), blue-tongue lizard (*Tiliqua scincoides*), and fence skink (*Cryptoblepharus virgatus*) may adapt, and even thrive, within the built environment (Koenig et al. 2002; Matthews et al. 2004; Garden et al. 2006). In contrast, urban-sensitive species (e.g., dunnarts, *Sminthopsis spp.*, antechinus, *Antechinus spp.*, geckos, and large reptiles) face significant dispersal, predator avoidance, and resource adaptation challenges within the human-dominated urban matrix. Consequently, these urban-sensitive species are restricted to the often isolated, remnant native vegetation patches that occur within the urban matrix or in the fringing peri-urban landscapes (How and Dell 1993, 1994; Tait et al. 2005; Garden et al. 2006).

Effective conservation in urban remnants requires scientific knowledge underpinning management decisions and on-ground actions. Much of the current available scientific knowledge is based on non-urban ecological research (Garden et al. 2006). In non-urban environments there is a strong global consensus that, regardless of the disturbance pressure, many native reptile and small mammal species depend more on structurally complex habitats, than compositionally diverse floristics (e.g., Bennett 1993; Southgate et al. 1996; Flemming and Loveridge 2003; Lenders and Daamen 2004; Spencer et al. 2005; Kanowski et al. 2006). Similar findings have also been reported for native species within Australian urban landscapes (e.g., Dufty 1994; Jellinek et al. 2004; White and Burgin 2004). Although, Fischer et al. (2003) and Jellinek et al. (2004) found that vegetation composition, in addition to structure, was important for reptile species in grazed and urban affected habitats. In contrast, Wilson et al. (1986) reported that within coastal heathlands of Victoria, several native rodent and dasyurid species displayed no floristic or vegetation structural preferences.

Such discrepancies in research findings, coupled with the distinct lack of urban-based ecological research (Garden et al. 2006), limit our ability to make generalisations about the habitat requirements of urban reptiles and small mammals, and hence management recommendations. This limitation is further compounded by differences in the type, rate and intensity of disturbances within urban landscapes compared to non-urban landscapes. A further complication is the large variation in habitat requirements among species (e.g., Jellinek et al. 2004; Monamy and Fox 2005), but also within species according to age. For example, although Fischer et al. (2003) found that four-fingered skinks (*Carlia tetradactyla*) responded to habitat structure and composition in grazing-affected landscapes, they also found that the relative importance of different attributes varied between juveniles and adults. Consequently, Australian urban conservation managers face significant uncertainty regarding the most appropriate management strategies for achieving long-term conservation outcomes for a diversity of native fauna species.

If native fauna diversity is to be conserved in the face of rapid urban expansion, it is vital that we understand the habitat requirements and sensitivities of species living within urban remnants. This requires understanding how both species composition and species richness are influenced by local-level habitat factors. However, local-level habitat management is often focussed on maintaining overall species diversity within the landscape by managing habitat patches so that current concomitant assemblages of native fauna are conserved before they become locally extinct, thereby avoiding the need for expensive reintroduction programs. Conserving native fauna in urban landscapes first requires understanding what local or *in-situ* factors are important for maintaining diverse assemblages of native fauna. This knowledge may then by used to inform urban conservation managers and planners about priority habitat management decisions and activities so that native fauna occupying urban habitat fragments are adequately conserved. This was the focus of our study.

This paper investigated correlations between the composition of native terrestrial reptile and small mammal fauna within urban habitat fragments and local-level (< 1 ha) habitat factors such as habitat structure, vegetation composition, fire and human disturbances. These habitat factors were considered ecologically important for the target species and also can be manipulated and managed by conservation managers. Investigations were based in urban habitat fragments located within the Brisbane City Council (BCC) local government area, where local government is responsible for setting within-patch management priorities and actions. We applied cluster analysis, multi-dimensional scaling and principal axis correlation to identify significant habitat attributes and examine their importance for reptile and small mammal assemblages. Mammal and reptile assemblages were analysed separately. Based on our findings, we discuss the implications for conservation management within Brisbane City.

4.3. METHODS

4.3.1. Study Area

The study focused on Queensland's capital city, Brisbane (153º2'S, 27ºE; area 1,220 km^2, population ~ 1 million) (Figure 4.1). Brisbane has a sub-tropical climate and is Australia's third largest and most rapidly growing capital city (Commonwealth of Australia 2003a). Since European settlement in the early 1800s approximately two-thirds of the original woody vegetation has been cleared for agricultural, industrial and/or urban development purposes

(Brisbane City Council 2001). More than 80% of this clearing was of lowland forests (< 100 m from sea level) resulting in the current, highly fragmented and isolated lowland remnant vegetation patches that vary in their internal condition and disturbance and management histories (Brisbane City Council 2001; Catterall and Kingston 1993). Of the remaining 33% remnant vegetation cover, about 20% is protected for conservation purposes, yet much of this area is concentrated in contiguous forest on the city's outskirts, particularly in the D'Aguilar ranges to the west (Brisbane City Council 2001).

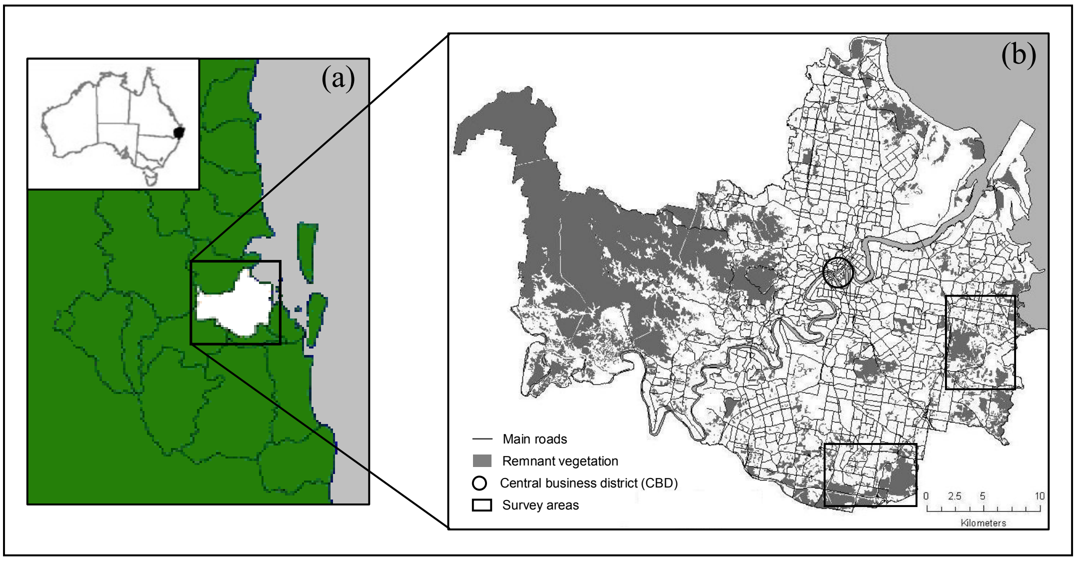


Figure 4.1. (a) Location of Brisbane City Council (BCC) local government area (LGA) on Queensland's south-east coast. (b) Map of BCC LGA showing the location of the survey areas in relation to the CBD. Remnant vegetation and major roads are also shown (Source data provided by Brisbane City Council).

Contemporary urban growth has concentrated along major transport networks to the north, south and south-west, with coastal wetlands and Moreton Bay to the east and mountain ranges to the west constraining urban expansion in these directions. Accordingly, the highest concentration of urban development pressure currently occurs in the lowland outer suburbs to Brisbane's south and southeast (Brisbane City Council 2001). Despite recent rapid urban expansion and subsequent habitat destruction and modification, Brisbane still supports high floristic and faunal diversity and endemism, boasting the highest diversity of native vertebrate species of any of Australia's capital cities (Queensland Museum 1995). However, in the face of rapid urbanisation, the continued persistence of this species diversity is uncertain. Our study focused on lowland remnant habitat fragments situated within 10-20 km of Brisbane's central business district, in the southern (Karawatha) and south-eastern (Burbank) suburbs (Figure 4.1).

4.3.2. Site Selection

We selected 59 sites located within Regional Ecosystem (RE) type 12.9-10.4, which is dominated by scribbly gum (*Eucalyptus racemosa*) woodland located on sedimentary rocks and sandy soils (Young and Dillewaard 1999). Although classified as "not of concern" by the Queensland Environmental Protection Agency (Queensland Environmental Protection Agency 2004), this RE is extensively cleared and fragmented within Brisbane City (Young and Dillewaard 1999). BCC spatial data and satellite imagery were used to select sites located on private and council owned land. Potential site locations were ground-truthed to assess their suitability. Site locations were considered unsuitable if they were: difficult to access; had been largely cleared or disturbed; patches were too small to fit a survey site; or, had a high likelihood of human interference to fauna survey equipment.

4.3.3. Site Design

Survey sites measured 45 m x 20 m and consisted of three parallel transects 10 m apart, along which fauna traps were placed and habitat assessments were conducted. Where possible, sites were positioned perpendicular to the landscape slope and a minimum of 10 m from patch edges. This distance was selected to concentrate on interior rather than edge and matrix habitats. Sites were also located at least 20 m from designated walking tracks and recreational areas.

4.3.4. Wildlife Surveys

Wildlife surveys were conducted over three consecutive nights, during fine weather conditions in the spring and summer of 2004 and 2005. Initial surveys (2004) were conducted at all 59 sites, yet repeat surveys (2005) occurred at only 51 sites, as eight sites were abandoned due to recent fire or substantial human interference. Native reptile and small mammal species were identified at each site using a combination of live-trapping, direct observation and trace survey methods, to maximise the probability of detecting the range of target species. Each site was surveyed using: 8 cage traps, 10 Elliott traps (2 sizes: 80 x 85 x 220 mm; 80 x 100 x 320 mm), 5 dry pit-fall traps (10 litre buckets), and 3 hair funnels. Opportunistic species identifications from scats, visual observations and vocalisations were also recorded.

Cage traps, Elliott traps and hair funnels were baited with the standard Australian native small mammal bait mixture (Menkhorst and Knight 2001). Pit-fall traps were unbaited. The dense vegetation and fallen woody debris at several sites prevented the use of drift-fences, which have previously been used with pit-fall traps to improve capture success (e.g., Mengak and

Guynn 1987; Crosswhite et al. 1999; Menkhorst and Knight 2001). Cage and Elliott traps were set and baited each afternoon, checked for captures each morning and closed during the day. Pit-fall traps remained open for the entire trap cycle (3 days and nights) and were checked for captures each morning and afternoon. Hair funnels were set at the beginning of a trap cycle and left undisturbed until collection at the end of the trap cycle. Scat collection, direct observations and vocalisation records occurred opportunistically throughout each site visit. At the point of capture, animals were identified to species-level using a relevant field guide (Menkhorst and Knight 2001; Wilson 2005), photographed, weighed, sexed (if possible) and immediately released. Scat and hair samples were identified *ex-situ* by one of two independent experts (Initial surveys: Michiala Bowen; Repeat surveys: Barbara Triggs).

4.3.5. Habitat Surveys

Habitat assessments were conducted at all sites between the initial and repeat wildlife survey periods. Twenty-five habitat variables (Table 4.1) were measured at each site, following a protocol similar to that detailed by Eyre et al. (2000). The number of large trees (DBH $\geq$ 40cm) within the site perimeter was visually counted from the centre of each site, and the total basal area was recorded using the Bitterlich variable radius method (*sensu* Mueller-Dombois and Ellenberg 1974). A visual appraisal of the total number of termite mounds, approximate weed cover, and the presence or absence of fire (e.g., tree scars, charcoal, ash) and human disturbances (e.g., litter, garden waste, sawn logs) were recorded over the duration of each site assessment.

Table 4.1. Habitat variables recorded at each site. Records for each variable are averaged/ totalled/calculated to produce a measurement across each site. * indicates variables not included in statistical analyses.

VARIABLE	DESCRIPTION	MEASUREMENT		
		Where	What	How
Acacia	Number of *Acacia*	1m either side of each transect	Total count	Visual
Allocasuarina	Number of *Allocasuarina*	1m either side of each transect	Total count	Visual
Banksia	Number of *Banksia*	1m either side of each transect	Total count	Visual
Bracken fern	Number of Bracken ferns	1m either side of each transect	Total count	Visual
Callistemon	Number of *Callistemon*	1m either side of each transect	Total count	Visual
Canopy cover	Amount of canopy cover (average of 30 readings)	Presence/absence reading taken every 5m along each transect	Percent cover	Gimballed Sighting Tube
Fire	Presence/absence of fire	Overall appraisal across site	Category	Visual
Ground cover complexity	Number of different types of ground cover (e.g., rocks, wood, coarse leaf litter)	$1m^2$ quadrat centred over transect, every 5m	Total count	Visual
* Ground cover coverage	Proportion of ground covered (i.e. not bare ground)	$1m^2$ quadrat centred over transect, every 5m	Percent cover	Visual
Ground cover depth	Depth (up to 40cm) of ground cover (average of 180 measurements)	$1m^2$ quadrat centred over transect, every 5m (6 measurements)	Average count	Standard 40cm ruler
Grass tree	Number of *Xanthorrhoea*	1m either side of each transect	Total count	Visual
Hollows	Number of pieces of fallen woody material with hollows	1m either side of each transect	Total count	Visual
Human Disturbance	Presence/absence of human disturbance (e.g., litter, sawn wood)	Overall appraisal across site	Category	Visual
Large trees	Number of large trees (DBH $\geq$ 40cm)	Sighted from centre of site	Total count	Visual and tape measure
Mid-storey Cover	Amount of mid-storey cover (average of 30 readings)	Every 5m along each transect	Percent cover	Gimballed Sighting Tube
Paperbark	Number of *Melaleuca*	1m either side of each transect	Total count	Visual
Soil Compaction	Number of "hits" (up to 20) (average of 180 measurements)	$1m^2$ quadrat centred over transect, every 5m (6 measurements)	Average count	Soil pen
Total basal area	Number of tree trunks > 1cm gap when viewed at eye level, 60cm from face	Sighted from centre of site	Total count (stems per m^2)	Bitterlich gauge

Termite mounds	Number of terrestrial termite mounds	Overall appraisal across site	Total count	Visual
Understorey Cover	Amount of understorey cover (average of 30 readings)	Presence/absence reading every 5m along each transect	Percent cover	Visual
Understorey density (total)	Total number of cells obscured by vegetation when a gridded canvas is viewed at waist height from a 5m distance	Every 5m along each transect	Average count	$1m^2$ canvas (gridded into $1cm^2$ cells)
* Understorey density (maximum)	Height of highest cell obscured by vegetation when a gridded canvas is viewed at waist height from a 5m distance	Every 5m along each transect	Average count	$1m^2$ canvas (gridded into $1cm^2$ cells)
* Understorey density (maximum 50%)	Highest cell at least 50% obscured by vegetation when a gridded canvas is viewed at waist height from a 5m distance	Every 5m along each transect	Average count	$1m^2$ canvas (gridded into $1cm^2$ cells)
Weed cover	Approximate proportion of site covered by weeds (0%, 25-50%, 50-75%, > 75%)	Overall appraisal across site	Percent cover	Visual
Wood Volume	Number, size and state of decay of fallen woody material	1m either side of each transect	Average volume (m^3)	Visual and calculated

The remaining 19 habitat variables were measured along each of the three transects and the measurements either totalled or averaged for the whole site. The total number of *Acacia spp.*, *Allocasuarina spp.*, *Banksia spp.*, *Pteridium esculentum*, *Callistemon spp.*, *Xanthorrhoea spp.* and *Melaleuca spp.* within one metre of the transect lines were recorded. Similarly, all fallen woody material within one metre of each transect line was tallied and categorised into five diameter classes (< 10 cm, 10-20 cm, 20-30 cm, 30-40 cm, > 40 cm), eight decay classes (recently felled, sound, bark peeling off, to 25% decay, 25-50% decay, 50-75% decay, >75% decay, debris), and three complexity classes (simple, complex, stump). These records were later used to calculate the relative volume (m^3) of fallen woody material for each site. The total number of pieces of fallen wood that contained hollows was also recorded.

The presence of canopy, mid-storey and under-storey cover was recorded every five metres along each transect using a gimballed sighting tube. Understorey density was also measured every five metres using an adaptation of methods described by MacArthur and MacArthur (1961) and Haering and Fox (1995). This involved a 1 m^2 screen, divided into 10 cm^2 grid cells, viewed at waist height from a two metre distance. Three understorey measurements were recorded: (1) the total number of grid cells obscured by vegetation, (2) the highest grid row obscured by vegetation (0cm–100cm), and (3) the highest grid row with at least one grid cell $\geq$ 50% obscured by vegetation. These three stratum records were later averaged to provide measures of average percent cover for each stratum and average understorey density at each site.

The percent of ground cover, number of ground cover types (e.g., coarse leaf litter, Casuarina needles, rocks, herbs), ground cover depth, and soil compaction were recorded every five metres within a 1 x 1 m quadrat along each transect. Percent and types of ground cover were assessed visually. A 40 cm ruler was used to record ground cover depth. Soil compaction was measured using a penetrometer (e.g., Fox et al. 1996) and counting the number of hits taken (up to 20 hits) to drive a weighted probe 20 mm into the soil. For both ground cover depth and soil compaction, six measurements were made every five metres. The records of percent ground cover, ground cover depth and soil compaction were later averaged for each site.

4.3.6. Statistical Analysis

Analyses were based on the species detected and habitat variables recorded at each site. Total species occurrence data were derived by combining the species detected from each wildlife survey method and during both survey periods. The overall species occurrence data were divided into two separate data sets: (1) native reptile species, and (2) native mammal species. Exotic species were not included in the data analysis.

Continuous habitat variables were initially compared using the Spearman rank correlation test in R version 2.1.0 (The R Development Core Team 2004). A correlation coefficient $\geq$ 0.7 was chosen to identify highly correlated habitat variables. Where variables were highly correlated, one variable was removed from the data set. The remaining variables were used for subsequent statistical analyses.

We applied a multivariate statistical approach using PATN Version 3.0.3 for Windows (Belbin 1994) to examine the relationships between species occurrence and site habitat characteristics. Similarities between sites in terms of species composition were first investigated using the Bray and Curtis index (Bray and Curtis 1957). Clusters were derived from the dataset using the flexible unweighted pair group using arithmetic averaging method (UPGMA) with a beta (ß) value of -0.1. The UPGMA clustering method is considered superior as it considers more than one species at any fusion (Wardell-Johnson and Williams 1996, Podani and Schmera 2006). At a beta value of -0.1, the UPGMA clustering method is space-dilating (cf. space-contracting), thereby creating even-sized groups and preventing the formation of a single large group (Belbin 1993, Wardell-Johnson and Williams 1996). We examined the acceptability of the resulting site groups and patterns using Semi-Strong Hybrid (SSH) multidimensional scaling ordination. The ordination was considered in conjunction with the minimum spanning tree (MST) and dendogram. The MST represents pair-wise associations between group objects. Its use is complementary to ordination and may be used practically to confirm or deny close relationships or identify trends in the data (Belbin 1993). The dendogram schematically displays the hierarchical clustering of groups. The number of site groups was adjusted until maximum congruence was obtained between the ordination, MST and dendogram, thereby indicating that the most robust site groupings had been attained.

Relationships between the intrinsic species occurrence and the ordination groups were examined using Principal Component Correlation (PCC) with 100 permutations. PCC uses multiple linear regression to fit selected variables to the ordination space, showing the direction of best fit and the correlation with that direction (r^2) (Belbin 1994). Monte-Carlo Attributes in Ordination (MCAO) procedure with 100 iterations was then used to test the reliability of these correlations with the ordination space by calculating the proportion of r^2 values that exceed the true r^2 in a given number of iterations (Belbin 1994). An MCAO value less than 5% was considered significant. Variables identified as having significant correlations with the species groups were then overlaid on the ordination space. This analysis procedure was repeated to investigate relationships between the extrinsic habitat variables and the ordination groups.

Species and habitat variables that were found to be highly correlated with the ordination were examined further using the Kruskal-Wallis statistic function in PATN to determine how well each variable discriminated between the ordination groups. The larger the Kruskal-Wallis statistic for a variable, the more significant that variable's contribution was to the differentiation of ordination groups. Significant variables according to the Kruskal-Wallis statistic ($P < 0.05$) were compared to MCAO values in order to test the strength of the relationships. We recognise, however, that the reliability of the Kruskal-Wallis statistic is proportional to the number of objects in the smallest group (Belbin 1993). As such, the Kruskal-Wallis statistic results are included, based on expert advice, as a comparative value to investigate the strength of MCAO results, but these values should be treated with caution.

4.4. RESULTS

4.4.1. Fauna Species Assemblages

A total of 27 native species (eight mammals, 19 reptiles) were identified from 333 trap nights (Table 4.2). An additional native mammal (*Melomys spp.*) was identified at two sites from scat and hair samples, but positive identification to the species level was not possible. Instead, the genus was included in the analysis of the mammal data set. In total, these represented 11 native family groups.

The most common mammal species was the common brushtail possum identified in 39 of the 59 sites (66%). However, it was considered a predominantly arboreal species (How and Kerle, 1995) and was not included in further analysis. The common dunnart (*Sminthopsis murina*) and the northern brown bandicoot (*Isoodon macrourus*) were the most commonly detected terrestrial mammal species (n = 8 sites each). Detections of the common planigale (*Planigale maculata*), common dunnart, and the yellow-footed antechinus (*Antechinus flavipes*) were of particular interest as these dasyurids are relatively uncommon in Brisbane's lowland habitats and are classified as significant species within the local government area (Brisbane City Council 2000). The fence skink (*C. virgatus*) and garden skink (*Lampropholis delicata*) were the two most common reptile species (n = 27 and n = 29 sites, respectively).

Larger-bodied and more cryptic reptile species such as snakes, lace monitors (*Varanus varius*) and dragons (e.g., bearded dragon, *Pogona barbata*, and eastern water dragon, *Physignathus lesuerii*) were identified at significantly fewer sites than small-bodied skink species (Table 4.2).

Table 4.2. Cumulative species list from wildlife surveys. Species are categorised as native/exotic mammals/reptiles and are listed alphabetically by family group and scientific name. Each species common name and the total number of sites at which they were detected are also shown.

	FAMILY GROUP	SCIENTIFIC NAME	COMMON NAME	SITES
Native Mammals	Dasyuridae	*Antechinus flavipes*	Yellow-footed antechinus	4
		Antechinus subtropicus	Subtropical antechinus	2
		Planigale maculata	Common planigale	2
		Sminthopsis murina	Common dunnart	8
	Muridae	*Melomys* spp.	Unknown species	2
		Rattus fuscipes	Bush rat	2
		Rattus lutreolus	Swamp rat	1
	Peramelidae	*Isoodon macrourus*	Northern brown bandicoot	8
		Perameles nasuta	Long-nosed bandicoot	2
Native Reptiles	Agamidae	*Diporiphora australis*	Tommy round-head	3
		Physignathus lesuerii	Eastern water dragon	1
		Pogona barbata	Bearded dragon	3
	Colubridae	*Dendrelaphis punctulata*	Common tree snake	1
	Elapidae	*Pseudechis porphyriacus*	Red-bellied black snake	1
	Gekkonidae	*Diplodactylus vittatus*	Eastern stone gecko	5
	Pygopodidae	*Lialis burtonis*	Burton's snake-lizard	1
	Pythonidae	*Morelia spilota*	Carpet python	1
	Scincidae	*Anamalopus verreauxii*	Verreaux's skink	1
		Calyptotis scutirostrum	Scute-snouted calyptotis skink	8
		Carlia foliorum	Tree-base litter-skink	5
		Carlia pectoralis	Open-litter rainbow skink	4
		Carlia vivax	Storr's rainbow skink	7
		Cryptoblepharus virgatus	Fence skink	27
		Ctenotus taeniolatus	Copper-tailed skink	1
		Eulamprus quoyii	Eastern water skink	1
		Lampropholis amicula	Secretive skink	4
		Lampropholis delicata	Garden skink	29
	Varanidae	*Varanus varius*	Lace monitor	3
Exotic Mammals	Canidae	*Vulpes vulpes*	Fox	3
	Leporidae	*Lepus capensis*	Brown hare	8
		Oryctolagus cuniculus	European rabbit	3
	Muridae	*Mus musculus*	House mouse	9
		Rattus rattus	Black rat	30
Exotic Reptiles	Gekkonidae	*Hemidactylus frenatus*	Asian house gecko	1

4.4.2. Habitat Variables Co-linearity

Correlation analysis revealed the three measures of understorey density were strongly correlated (≥ 0.7). We eliminated two variables and retained total understorey density. Although not highly correlated with other variables, average ground cover was also removed from the data set as it showed very little variation between sites with 51 sites having an average cover > 90%.

4.4.3. Reptile Groups and Habitat Associations

For reptiles, four groups of sites (Figure 4.2a) yielded the lowest ordination stress level (stress = 0.183) and maximum congruence between the ordination, minimum spanning tree and dendogram. The two most commonly captured reptile species, *C. virgatus* and *L. delicata*, dominated Group 1 sites (n = 24), being identified at 92% and 67% of sites respectively (Figure 4.3a). This indicates a high degree of overlap in the distribution of these species within Group 1. However, these skink species showed little or no overlap in the 12 and 13 sites that respectively comprised Groups 2 and 3. Group 2 sites were characterised by the ubiquitous presence of *L. delicata* (detected in all sites) and the absence of *C. virgatus* (Figure 4.3a). The presence of *C. virgatus* rather than *L. delicata* dominated Group 3 (38% and 8%, respectively) (Figure 4.3a). Species richness also varied between groups, with Group 3 containing the highest overall native species richness (12 species), more than 40% greater species richness than Group 2 (Figure 4.3a). Group 4 represented the fewest sites (n = 10) and was characterised by a total absence of native reptile species.

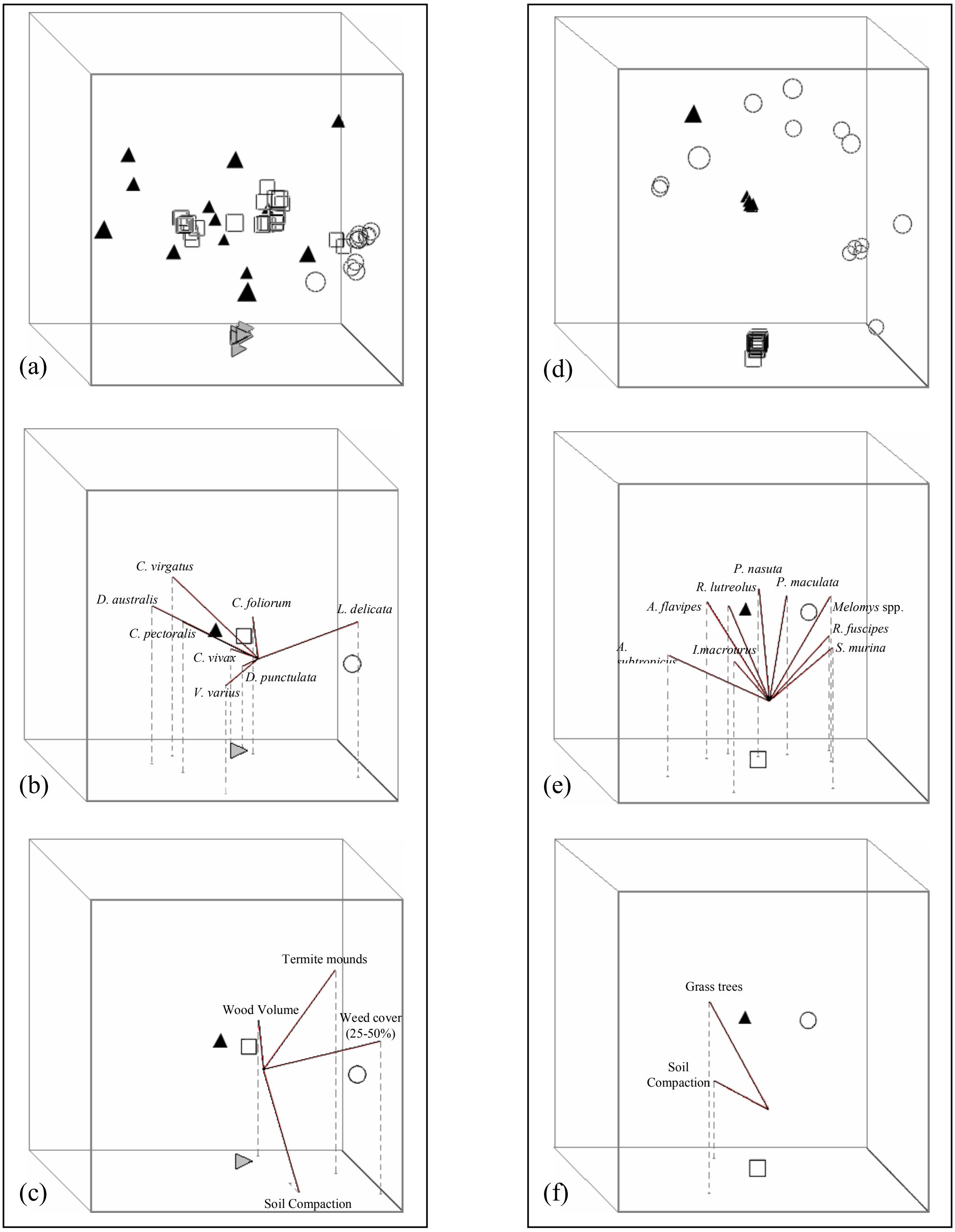
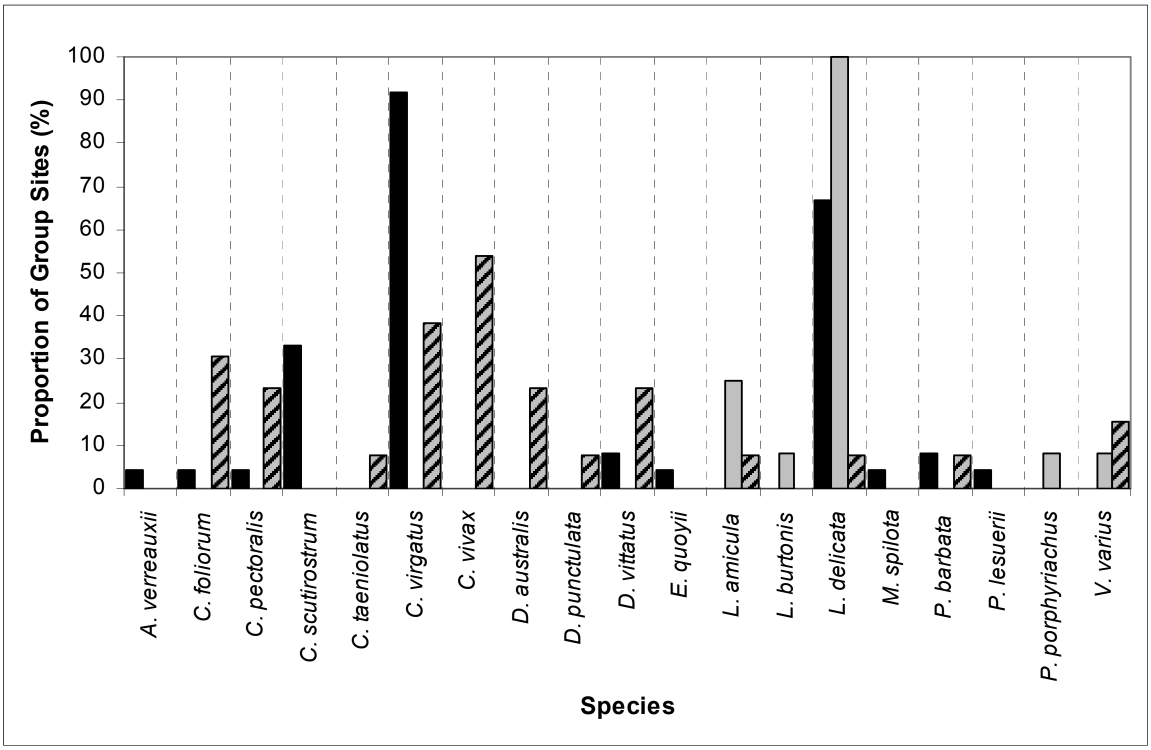


Figure 4.2. SSH MDS ordinations for reptiles (a-c) (stress = 0.1832) and mammals (d-f) (stress = 0.1283). (a) and (d) show cluster ordination of sites according to similarities in species composition; (b) and (e) show significant species overlaid; (c) and (f) show significant explanatory habitat variables overlaid. Group centroids only are shown in (b), (c), (e) and (f). For all ordinations: □ Group 1; ○ Group 2; ▲ Group 3; ▷ Group 4 (reptiles only).

(a)



(b)

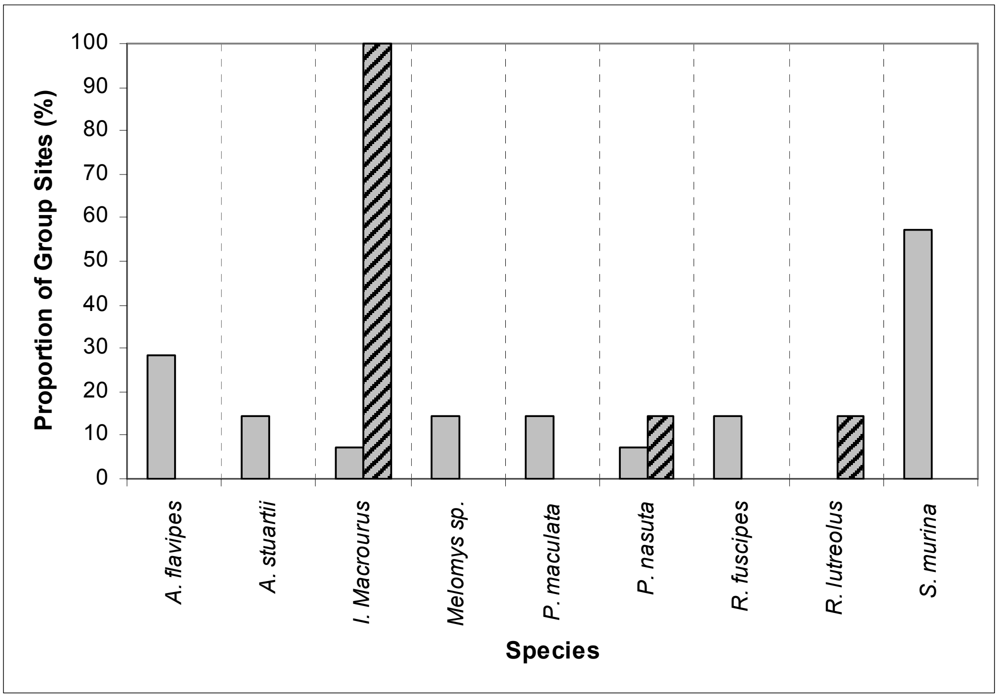


Figure 4.3. Relative species compositions for each of the reptile and mammal site Groups, showing the proportion of sites within each Group at which species were detected. (a) Reptile species compositions for Groups 1-3; Group 4 not shown as no native reptile species were detected at sites within this group; Group 1=24 sites, Group 2=12 sites, Group 3=13 sites, Group 4=10 sites. (b) Mammal species compositions for Groups 2 and 3; Group 1 not shown as no native mammal species were detected at sites within this group; Group 1=38 sites, Group 2=14 sites, Group 3=7 sites. For both (a) and (b): ■ Group 1; □ Group 2; ▨ Group 3.

MCAO reported eight native reptile species that were highly correlated with the ordination of reptile groups: *Carlia foliorum*, *C. vivax*, *C. virgatus*, *Dendrelaphis punctulata*, *Diporiphora australis*, *L. delicata*, *Carlia pectoralis* and *Varanus varius* (Figure 4.2b). Of these species, the presence of *C. virgatus*, *L. delicata* and *C. vivax* explained the largest amount of variation in the ordination (r^2 = 0.52, 0.67, and 0.35 respectively) and were highly significant at $P <$ 0.001 (Table 4.3). *C. foliorum*, and *D. australis* were less significant ($P < 0.05$). Although *C. pectoralis*, *D. punctulata* and *V. varius* were highly correlated with the ordination, these species were not found to be statistically significant (Table 4.3). The results further suggest that *C. scutirostrum* and *L. amicula* discriminate well between the groups ($P < 0.01$ and $P <$ 0.05, respectively), yet MCAO indicated these species had a low correlation with the ordination groups (Table 4.3).

Four habitat variables: soil compaction, total number of termite mounds, wood volume, and weed cover (25-50%), were significantly correlated with reptile groups (Figure 4.2c). Of these variables, only soil compaction and weed cover (25-50%) discriminated well between the groups according to the Kruskal-Wallis statistic (Table 4.3).

4.4.4. Mammal Groups and Habitat Associations

Three statistical groups of native mammal species (Figure 4.2d) produced the best ordination fit (stress = 0.1283) and maximum congruence between the ordination, minimum spanning tree and dendogram. Group 1 was comprised of the 38 sites at which no native mammal species were detected, indicating that native mammal species were undetected at the majority of the 59 sites. Comparatively, Groups 2 and 3 were dominated by the Dasyuridae and

Peramelidae families respectively. Native rodent species (Family: Muridae) were represented in Group 2 and Group 3 (Figure 4.3b). All native mammal species were correlated with the ordination (Figure 4.2e) and, with the exception of northern brown bandicoots, explained between 17% and 30 % of the group variation (Table 4.3). Northern brown bandicoots explained more than 50% of the group variation ($r^2 = 0.505$), being found at all sites comprising Group 3 (Figure 4.3b). The statistical significance of all but one species (long-nosed bandicoot, *Perameles nasuta*) was supported by the Kruskal-Wallis statistic (Table 4.3). Both northern brown bandicoots and common dunnarts had the highest discrimination between groups ($P < 0.001$).

The number of grass trees (*Xanthorrhoea spp.*) and soil compaction were the only habitat variables that were significantly correlated with the mammal ordination (Figure 4.2f). However, the associated Kruskal-Wallis values indicated that neither of these variables discriminated well between the groups (Table 4.3).

Table 4.3. Relative associations between ordination Groups and native reptile/mammal species and habitat variables. PCC (r^2), MCAO (%) and chi-squared (derived from KW statistic) are shown. Significant MCAO variables and values are indicated in bold font. Chi-squared significance is indicated by: * 0.05 > *P* > 0.01; ** 0.01 > *P* > 0.001; *** *P* < 0.001; n/s = not significant. The ordination Group with which each species and habitat variable was most highly correlated is also shown.

Reptile Species	PCC (r^2)	MCAO (%)	χ^2	Group	Mammal Species	PCC (r^2)	MCAO (%)	χ^2	Group
A. verreauxii	0.02	80	n/s	1	***A. flavipes***	0.3	**1**	**	2
C. foliorum	0.03	**0**	*	3	***A. subtropicus***	0.3	**1**	*	2
C. pectoralis	0.19	**1**	n/s	3	***I. macrourus***	0.51	**0**	***	3
C. scutirostrum	0.03	72	**	1	***Melomys spp.***	0.2	**3**	*	2
C. taeniolatus	0.15	11	n/s	3	***P. maculata***	0.29	**0**	*	2
C. virgatus	0.52	**0**	***	1	***P. nasuta***	0.21	**4**	n/s	3
C. vivax	0.35	**0**	***	3	***R. fuscipes***	0.22	**2**	*	2
D. australis	0.32	**0**	*	3	***R. lutreolus***	0.17	**4**	*	3
D. punctulata	0.18	**0**	n/s	3	***S. murina***	0.18	**2**	***	2
D. vittatus	0.16	9	n/s	3					
E. quoyii	0.04	54	n/s	1					
L. amicula	0.04	50	*	2					
L. burtonis	0.06	19	n/s	2					
L. delicata	0.67	**0**	***	2					
M. spilota	0.04	46	n/s	1					
P. barbata	0.15	5	n/s	1					
P. lesuerii	0.01	90	n/s	1					
P. porphyriacus	0.04	53	n/s	2					
V. varius	0.15	**1**	n/s	3					
Habitat Variable	PCC (r^2)	MCAO (%)	χ^2	Group	Habitat Variable	PCC (r^2)	MCAO (%)	χ^2	Group
Acacia	0.01	91	n/s	3	*Acacia*	0.03	66	n/s	1
Allocasuarina	0.07	31	*	1	*Allocasuarina*	0.01	95	n/s	3
Banksia	0.08	18	n/s	2	*Banksia*	0.08	15	n/s	3
Bracken fern	0.05	60	n/s	3	Bracken fern	0.05	35	n/s	1
Callistemon	0.04	50	n/s	1	*Callistemon*	0.1	29	n/s	3
Canopy cover	0.07	32	n/s	1	Canopy cover	0.03	65	n/s	2
Fire	0.01	95	n/s	4	Fire	0.04	54	n/s	3
Ground cover complexity	0.06	34	*	4	Ground cover complexity	0.03	60	n/s	3
Ground cover depth	0.02	82	n/s	2	Ground cover depth	0.05	40	n/s	1
Human disturbance	0.05	40	*	2	Human disturbance	0.06	36	n/s	3
Hollows	0.03	66	n/s	3	Hollows	0.03	65	n/s	3
Large trees	0.05	44	n/s	3	Large trees	0.04	47	n/s	2
Melaleuca	0.1	16	n/s	2	*Melaleuca*	0.13	10	n/s	3
Mid-storey cover	0.07	22	n/s	2	Mid-storey cover	0.02	76	n/s	3
Soil compaction	0.21	**0**	*	4	**Soil compaction**	0.17	**4**	n/s	3
Total basal area	0.03	62	**	1	Total basal area	0.05	36	n/s	2
Termite mounds	0.14	**4**	n/s	3	Termite mounds	0.06	35	n/s	2
Under-storey cover	0.02	71	*	3	Under-storey cover	0.02	72	n/s	2
Under-storey density	0.03	71	n/s	3	Under-storey density	0.04	48	n/s	3
Weed cover (0%)	0.12	12	**	3	Weed cover (0%)	0.03	77	n/s	2
Weed cover (≤ 25%)	0.08	17	n/s	4	Weed cover (≤ 25%)	0.02	84	n/s	3
Weed cover (25-50%)	0.15	**3**	*	2	Weed cover (25-50%)	0.1	9	n/s	3
Weed cover (50-75%)	0.06	23	n/s	4	Weed cover (50-75%)	0.01	63	n/s	1
Wood volume	0.15	**2**	n/s	3	Wood volume	0.1	13	n/s	2
Xanthorrhoea spp.	0.03	56	n/s	1	***Xanthorrhoea spp.***	0.21	**2**	n/s	2

4.5. DISCUSSION

This study considered a range of local-level environmental factors and their correlation with native terrestrial reptile and small mammal species compositions within fragmented urban bushland habitats. Our results initially appear to imply that both vegetation composition (weed cover and grass trees) and habitat structure (termite mounds, wood volume, and soil compaction) are important for native reptile and small mammal species. However, when previous research, and species behaviours and life history traits are examined, it seems likely that species in this study were responding to the structural role fulfilled by weed cover and grass trees rather than the compositional or floristic role. Therefore, we conclude that at the local-level, habitat structural complexity is more important than vegetation composition for the occurrence of terrestrial, native reptile and small mammal species in Brisbane's lowland remnant habitat fragments.

This overall result is consistent with several other studies of various fauna species (including birds, reptiles, and mammals) living within natural and disturbed (non-urban) habitats (e.g., Grover and Slater 1994; Catterall et al. 1998; Scott et al. 1999; Webb and Shine 2000; Tait et al. 2005; Vesk and Mac Nally 2006). Many of these and similar studies indicate distinct species-specific responses to particular habitat attributes, yet it was not possible in our study to identify species-specific relationships because of the low detection rate of many species in several sites. The low detection of several species may potentially be explained by localised species declines due to urbanisation and its associated disturbances, and/or due to the cryptic nature of habitat use, dietary preferences, and seasonal population fluctuations resulting in a high degree of false-absences. Additional surveys conducted over a longer time period are likely to have improved detection rates, added certainty to our results, and provided more information on the habitat preferences of individual species. However, such long-term surveys were not possible in this study due to time and resource constraints. Consequently, our analysis focused on the influence of habitat characteristics on native reptile and small mammal species compositions rather than individual species occurrences.

4.5.1. Reptiles

Reptile captures were dominated by two main species, *L. delicata* and *C. virgatus*. Both species are also found throughout the urban matrix, although *C. virgatus* apparently more so than *L. delicata*, (J. Garden pers. obs.). The recorded dominance of these species during fauna surveys played a significant role in determining the subsequent reptile groups in the PATN analysis, indicating that these skink species may influence the composition of skink assemblages within urban patches. Similar reptile inter-species interactions were reported by Fischer et al. (2003) who found that *C. tetradactyla* more frequently inhabited sites occupied by at least two other small reptile species. The possibility of inter-species interactions between other small reptile species and *L. delicata* and/or *C. virgatus* is a hypothesis that warrants further investigation in order to clearly delineate existing relationships and examine the influence of inter-species interactions relative to habitat suitability.

Termite mounds and fallen woody material

The significance of termite mounds and fallen woody material is likely explained by the habitat and resource requirements of reptiles. Reptiles are ectothermic and so are dependent on habitat attributes that enable them to regulate their body temperature to achieve optimal performance, which is essential for foraging, breeding and predator-avoidance behaviours (Huey 1991; Bauwens et al. 1996; Vitt et al. 1998; Burrow et al. 2001; Singh et al. 2002; Lenders and Daamen 2004). Termite mounds and fallen woody material both provide suitable basking locations, habitats for prey species, and numerous nesting and refuge niches. It is not surprising, therefore, that these structural attributes were most strongly associated with sites dominated by the presence of sun-loving reptile species such as *C. vivax*, *Ctenotus taeniolatus*, and *D. australis* (Group 3).

The association of reptiles with termite mounds is consistent with previous studies, both in Australia and elsewhere. For example, a lizard species (*Metroles cuneirostris*) from the Namib Desert of coastal Namibia, was found to commonly utilise newly formed termite mounds as foraging locations (Murray and Schramm 1987). The use and importance of termite mounds as refuge locations has also been reported for different reptile species, such as inactive monitors (*Varanus bengalensis*) in Sri Lanka (Wikramanayake and Dryden 1993) and frillneck lizards (*Chlamydosaurus kingii*) in Australia's Northern Territory (Griffiths and Christian 1996). Griffiths and Christian (1996) particularly demonstrated the importance of termite mounds as refuges by showing that *C. kingii* individuals who utilised termite mounds to avoid high intensity fires had a 100% survival rate, compared to the increased mortality and injury of individuals that sought refuge in tree canopies.

As supported by previous research, the significance of fallen woody material is likely to provide similar resources as termite mounds for a suite of small-bodied, reptile species (e.g., Smith et al. 1996; Fischer et al. 2004; Jellinek et al. 2004). Other ground-level structural attributes, such as ground cover/leaf litter (e.g., Burrow et al. 2001; Singh et al. 2002) and bush-rocks (e.g., Schlesinger and Shine 1994; Webb and Shine 2000; White and Burgin 2004) have also been reported to provide suitable basking, shelter and foraging opportunities for various native terrestrial reptile species (e.g., snakes, skinks, agamids, and geckos). These structures are naturally occurring yet, in degraded urban remnants, human-introduced materials such as discarded metal and wood, and even old car bodies may work equally well (J. Garden pers. obs.).

Weed Cover

The occurrence of native reptiles was positively associated with a moderate amount of weed cover (25-50%). Weediness was particularly associated with Group 2 which was characterised by comparatively 'secretive' reptile species such as *L. delicata*, *L. amicula,* and Burton's snake-lizard (*Lialis burtonis*). Although all species require habitats that facilitate thermoregulation, Group 2 species appear to tolerate a moderate amount of certain habitat disturbances (as indicated by weediness), so long as adequate vegetation cover is available either to facilitate thermoregulation or to provide rapid refuge from predators. This is consistent with Bragg et al. (2005) who found that *L. delicata* was more likely to inhabit forested areas, which had more leaf litter, ground and shrub cover than the adjoining open habitat of regenerating mine-disturbed areas. Furthermore, in our study *L. delicata*, unlike *C. virgatus*, were more likely identified from pit-fall captures than direct observations, a trend that is indicative of the more cryptic nature of *L. delicata*. Similar capture trends were noted by Singh et al. (2002) for *L. delicata* and *C. virgatus* surveyed in contiguous forest near Brisbane. *L. burtonis* and *P. porphyriacus* were also identified from direct observations as pit-fall traps were not large enough to trap these larger-bodied species. However, upon detection, these species were observed to actively seek refuge within relatively dense thickets of lower-stratum vegetation.

The positive association, therefore, between the occurrence of certain reptile species and a moderate amount of weed cover is most likely due to the shelter provided by low, weedy vegetation rather than the weed species composition *per se*. These findings support those of Fischer et al. (2003) who noted that juvenile and some adult *C. tetradactyla* were found in moderate to highly weed-infested habitats, concluding that these species are able to tolerate a certain degree of habitat disturbance and potentially benefit from the associated structural changes. Other researchers have similarly commented on the importance of lower-stratum vegetation cover for supporting important reptile prey species and also for the safe shelter provided from predators whilst foraging and dispersing (e.g., Burrow et al. 2001; Fischer et al. 2003).

Although it appears that some reptile species respond positively to a certain degree of weed cover, Hadden and Westbrooke (1996) and Jellinek et al. (2004) reported that overall reptile species richness was negatively associated with increased weediness. Our results provide some agreement with these previous findings. Moderate weed cover was most associated with Group 2 which, compared to Groups 1 and 3, also had the lowest overall native species richness. Further, the highest amount of weed cover (50-75%), although not significantly correlated with the ordination, appeared to be associated with Group 4 (no reptiles detected). Jellinek et al. (2004) discuss the possible influence of weediness and time since isolation, commenting on smaller and older remnants being more likely to be dominated by weeds than larger and younger remnants. This may potentially be critical for effectively managing reptile species compositions investigated in the current study. As weed invasions are indicative of disturbed habitats, it seems likely that, as highlighted by Jellinek et al. (2004) and suggested by the current study's findings, certain reptile species are sensitive to habitat disturbances and so will respond negatively to even a low amount of weed cover. Conversely, some reptile species are able to tolerate a certain degree of habitat disturbance and may even benefit from the cover provided. It is therefore difficult to make generalisations of the importance of weeds for all reptiles.

Soil Compaction

Increased soil compaction was characteristic of habitats in which few or no native reptile species were detected, with the hardest soils occurring at sites at which no native reptile species were detected (Group 4). There are two possible explanations for this finding. The first directly implicates soil compaction and considers its impact on species behaviours, whereas the second considers indirect implications of soil compaction, its associated disturbances on vegetation structure, and the resulting influence on reptile species. Soil compaction was least associated with Group 1, indicating that reptiles in this group occur more frequently in habitats with soft soils. This is consistent with the two fossorial skink species identified, *Calyptotis scutirostrum* and *Anomalopus verreauxii*, which were found only at sites within this group. Due to their burrowing behaviour, these species do not inhabit or persist in areas with hardened soils where burrowing is difficult.

Soil compaction also has a significant influence on vegetation growth and regenerative ability (Amrein et al. 2005; Bassett et al. 2005). Increased soil compaction is often a result and a consequence of decreased vegetation cover (e.g., Groves and Keller 1983; Hadden and Westbrooke 1996). This cyclic condition is intensified by external disturbances which directly compact the soil and destroy ground cover vegetation. Hence, habitats with compacted soils are also likely to be indicative of highly disturbed habitats. Inappropriate fire regimes (including arson fires) and aesthetic clearing which decrease vegetation cover and do not promote vegetation regeneration are likely to increase soil compaction and, in turn, make it more difficult for plant species to regenerate (Amrein et al. 2005). As a result, the habitat structure and soil condition degrades, negatively impacting on various reptile species dependent on structurally complex habitats. Similarly, off-track trampling within urban bushland habitats compacts soil and destroys or degrades ground cover vegetation, making these habitats less suitable for reptiles.

4.5.2. Mammals

Mammal species were found to be influenced primarily by habitat structure rather than vegetation composition. This is consistent with previous studies (e.g., Bennett 1993; Haering and Fox 1995; Monjeau et al. 1998; Vernes 2003; Monamy and Fox 2005) which concluded that vegetation structure, rather than vegetation composition, was more important for small mammal occurrence, although species-specific responses to various aspects of structure were evident. For instance, small-bodied mammals, such as dasyurids, are likely to be more capable of moving rapidly through dense undergrowth, whereas large-bodied mammals such as kangaroos are impeded by dense midstorey cover, but are less likely to be affected by dense understorey and ground cover. Comparatively, large areas of dense undergrowth may present a significant locomotor (and escape) obstacle for medium-bodied mammals, such as bandicoots, and may also inhibit bipedal vigilance behaviours (Garden 2000). Vernes (2003) found that northern bettongs (*Bettongia tropica*) avoided areas of dense ground cover, particularly dense cover within 0.5 m above ground, which approximates the height of an adult bettong. Keiper and Johnson (2004) similarly reported that short-nosed bandicoots (*Isoodon obesulus peninsulae*) in north Queensland forests avoided habitats with a tall, dense grass understorey. Such a response is also likely to be true for bandicoots in urban landscapes and so these animals may actively select habitats that are structurally complex, yet are not vegetatively dense. Therefore, as suggested by previous bandicoot studies (e.g., Dufty 1994; Scott et al. 1999; Chambers and Dickman 2002), optimal habitats for Peramelidae are structurally complex and encapsulate a mosaic of open foraging areas and denser shelter sites that are not so dense as to impede locomotor ability.

Grass Trees

Grass trees also appear to be an important factor for the occurrence of native mammal species. This finding supports previous studies that have identified grass trees as important structural elements for several Australian small terrestrial mammal species. Vernes and Pope (2001), for instance, found that prior to fires, almost half the number of *B. tropica* nests were located in dense vegetation cover such as the skirts of grass trees. Spencer et al. (2005) similarly found that native bush rats (*Rattus fuscipes*) responded negatively to the decrease in grass tree cover following fires. Likewise, Lunney (1995, p. 651) commented that 'cover-seeking' *R. fuscipes* prefer habitats with dense ground and understorey cover – a structural requirement that would be partially filled by the presence of grass trees in the understorey layer. Grass trees have also been documented as providing important nesting habitats for small dasyurid species such as the common dunnart (Fox 1995) and the yellow-footed antechinus (Marchesan and Carthew 2004). Our results support these previous findings, in that the presence of grass trees was most strongly associated with Group 2, which was dominated by dasyurids and *R. fuscipes*.

Soil Compaction

Soil compaction, as for reptiles, was also associated with mammal species occurrences. Like reptiles, mammal species also appear to avoid habitats with very hard soils and the reasons may be similar. Firstly, certain mammal species such as some dasyurids may nest in terrestrial burrows (Woolley 1989; J. Garden pers. obs.) and harder soils would inhibit their ability to burrow. However, unlike the reptile results, certain mammal species appear to respond positively to compacted soils. Mammal Group 3 was most strongly associated with moderately compacted soils rather than very hard soils. This group was dominated by the Peramelidae (bandicoot species). Bandicoots are omnivorous and most commonly feed on subterranean invertebrate prey and plant structures such as roots and hypogeal fungi (Vernes 2003; Keiper and Johnson 2004), a behaviour that is obvious from the conical diggings left from foraging (Triggs 1996; J. Garden pers. obs.). Their ability to obtain these food resources is possible due to specialised strong claws on their forefeet which make them powerful diggers. This morphological trait may provide them with an advantage over other species in enabling them to penetrate into harder soils and therefore utilise habitats with increased soil compaction. However, if soils become too hard, vegetation cover decreases (Amrein et al. 2005), resulting in decreased soil moisture and microbial activity, and hence less food availability and increased energy output (for foraging). Structurally complex habitats are likely to occur where there is low to moderately compacted soils influencing vegetation floristics and cover. In addition, although bandicoots appear to prefer structurally open habitats for foraging, they have also been reported to require structurally dense habitats for diurnal shelter (e.g., Dufty 1994; Southgate et al. 1996; Chambers and Dickman 2002; Vernes 2003). Therefore, as soils harden and vegetation cover decreases, important food resources decrease as does critical vegetation cover. However, in the urban landscape, Dufty (1994) reported that the eastern-barred bandicoot (*Perameles gunii*) used a range of natural and artificial structures as shelter sites, implying a certain degree of disturbance tolerance, including moderately compacted soils.

4.5.3. Management Implications

Understanding the local-level habitat requirements of native fauna species is an essential component of successful urban biodiversity conservation. Our findings indicate that, for native terrestrial reptiles and small mammals, maintaining structurally complex habitats takes priority over managing local-level vegetation composition. This is perhaps best achieved by managing and minimising disturbances that degrade habitat structure. Based on our findings, we suggest three main areas of consideration for urban habitat management at the local-level.

First, fire is an important component of all habitats, both within and external to urban areas. In urban areas, fire is used as a management tool to protect people and their assets. The main aim of planned management fires is to reduce natural fuel loads, such as fallen woody material and ground/leaf litter cover, so that un-managed fires are less intense and more easily contained. Consequently, managed burns in urban areas are often at a lower intensity but higher frequency than naturally occurring fires in non-urban habitats. Such fire regimes remove, decrease and/or degrade important habitat structural elements and, alter the regeneration time and structural complexity of habitat fragments. As a result, these urban management practices have varying short- and long-term impacts on native species. Whereas certain reptile and mammal species may benefit from the open structure immediately following fires, others may respond more to the moderate cover during regeneration, whereas some species may not be found in an area until vegetation density, habitat structure and ground cover increases long after a fire (e.g. Hannah et al. 1998; Fox et al. 2003). In non-urban landscapes, species exhibit varying rates of recolonisation into burnt areas (Fox 1982), yet the likelihood of species recolonising burnt habitats in urban areas is additionally dependent on their ability to traverse the urban matrix in order to access burnt areas. Therefore, it is important that fire regimes are closely examined in terms of their short- and long-term impacts on vegetation, habitat structural attributes and associated fauna species assemblages. It is also important that this knowledge is incorporated into mosaic burning practices and fire frequency management.

Second, it is critical that practices such as aesthetic clearing, habitat (e.g. bush-rocks and wood) and plant removal, garden waste dumping, and off-track trampling are discouraged. Such habitat misuse increases the disturbance levels and so facilitates weed invasions, decreases vegetation structure and also degrades other structural attributes such as termite mounds and fallen woody material, thereby negatively impacting native species richness. We propose that priority habitat fragments are specifically targeted by local habitat managers. This includes areas that currently support high native species richness and also those that are being managed to restore species richness, particularly when these areas have a high human use (e.g. walking tracks, picnic facilities). It is important in these priority areas that existing controls on human misuse are enforced.

Third, community revegetation groups are vital for efficiently managing and monitoring habitat fragments for the benefit of native fauna richness. However, such groups should first consider the structural rather than compositional habitat attributes of their focal patch. Habitat regeneration actions that commonly begin by replacing weedy plants with native plants would be better advised to assess the overall weediness and structure of habitats first. If weed cover is relatively low, resources may be better focussed on enhancing habitat structural complexity. This could be achieved by: conserving and planting large eucalypt tree species that provide important fallen woody material and leaf litter cover that enhance vertical structure; promoting vegetation structural complexity in mid- and under-story layers; and even introducing artificial structures that may act as surrogates for natural basking, shelter and nesting structures used by native terrestrial mammal and reptile species.

Finally, although it is important to understand the habitat requirements of individual fauna species, when the goal is conserving, restoring or increasing native fauna biodiversity, habitats must not be managed based on the requirements of a single species. This is often difficult due to the variety of species-specific responses and the cryptic nature of many native Australian terrestrial fauna. A potential way to address this challenge and facilitate effective management of habitats for multiple species is to construct ecological species profiles (Opdam et al. 2002). Such profiles enable various species to be categorised based on similarities in their habitat requirements and disturbance responses. The resulting ecological species profiles may then be used to guide habitat management decisions, such as priority areas, actions and funding allocation, and facilitate the long-term conservation of multiple species. In addition, long-term monitoring of reptiles and ground-dwelling mammals in remnant habitat fragments, and associated adaptive management strategies, is recommended in order to minimise inherent problems of short-term studies such as, false-absence records and seasonal fluctuations in species diversity and abundance.

(a) Eastern stone gecko (*Diplodactylus vittatus*)
- Family: Gekkonidae
- Insectivorous, nocturnal gecko.
- Average size: SVL 50 mm.
- Distribution: Queensland, New South Wales, Victoria, South Australia.
- In Brisbane: restricted to suitable remnant bushalnd habitat in peri-urban landscapes.

This photo shows an adult male eastern ston gecko on my hand.

(b) Lace monitor (*Varanus varius*)
- Family: Varanidae
- Carnivorous, diurnal lizard.
- Average size: Total length 2.1 m.
- Distribution: Queensland, New South Wales, Victoria, South Australia.
- In Brisbane: common in remnant bushland in peri-urban landscapes.

In this photo an adult lace monitor is taking refuge on a tree trunk.

(c) Verreaux's skink (*Anomalopus verreauxii*)
- Family: Pygopodidae
- Nocturnal, fossorial skink.
- Average size: SVL 185 mm.
- Distribution: Queensland and New South Wales.
- In Brisbane: common in gardens and bushland, particularly in peri-urban landscapes.

In this photo I am holding a juvenile Verreaux's skink – note shortened forelimbs, and hindlimbs reduced to stump

(d) Copper-tailed skink (*Ctenotus taeniolatus*)
- Family: Scincidae
- Insectivorous, diurnal skink.
- Average size: SVL 80 mm
- Distribution: Queensland, New South Wales, Victoria.
- In Brisbane: common in parks and bushland throughout, but particularly in peri-urban landscapes.

This photo shows an adult copper-tailed skink on my hands.

Plate 5. Gecko, monitor, and skinks: (a) Eastern stone gecko; (b) Lace monitor; (c) Verreaux's skink; (d) Copper-tailed skink.

Chapter 5

WHAT'S MORE IMPORTANT FOR WILDLIFE IN FRAGMENTED URBAN LANDSCAPES – LOCAL, PATCH, OR LANDSCAPE-LEVEL INFLUENCES? A REPTILE AND SMALL MAMMAL CASE STUDY FROM SOUTHEAST QUEENSLAND, AUSTRALIA

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5.1. ABSTRACT

As urban areas continue to expand, and fragment and replace natural ecosystems worldwide, the effective conservation of wildlife populations in fragmented urban and peri-urban landscapes is becoming increasingly important. However, the processes enabling wildlife to persist in urban areas are not well understood. We addressed the question: What is more important for native species richness in fragmented urban landscapes – local (< 1 ha), patch (1

– 100s ha), or landscape-level (100s – 1000s ha) influences? To answer this question we studied native terrestrial reptile and small mammal species assemblages within lowland remnant vegetation fragments of Brisbane City, south-east Queensland (Australia). Species richness and local-level habitat attributes were surveyed at 51 field sites for two repeat surveys over two years. Patch and landscape-level attributes were measured at increasing radial extents (500– 5000 m) around each survey site. Generalised linear modelling and hierarchical partitioning were used to determine the importance of the amount of forest habitat and its configuration, relative to patch size and shape, and local vegetation composition and structure. We found that influences at all levels were important for both reptile and mammal species richness. Reptile species richness was influenced foremostly by the amount of forest habitat and its configuration at the landscape-level, and weed cover and soil compaction at the local-level. The key factors influencing mammal species richness were the amount of forest and rural/low density urban habitat at the landscape-level, followed by habitat composition at the local-level, and patch size and shape at the patch-level. These findings highlight the importance of adopting a hierarchical landscape perspective for the conservation of urban wildlife.

Key Words: Conservation, Management, Akaike Information Criteria, Relative importance, Brisbane.

5.2. INTRODUCTION

The process of urbanisation is the most rapidly expanding and enduring form of anthropogenic landscape change. This process significantly alters biotic, abiotic, and physical attributes of the natural environment (Botkin and Beveridge 1997) resulting in transformed and degraded ecosystems that may never return to their original ecological state (Lugo 2002). The primary impacts of urban development on natural ecosystems are the loss, fragmentation, and degradation of natural habitats, which have significant ramifications for native fauna populations (Baskin 1998; Wilcove et al. 1998; Fahrig 1997; Marzluff and Ewing 2001; Faulkner 2004). Whereas certain species may thrive in the modified urban landscape, such as exotic species and some native generalist species, many native species are negatively impacted by urbanisation (Baskin 1998; Garden et al. 2006). Native fauna populations fragmented by urban development undergo population declines and localised extinctions often long after development occurs (Tilman et al. 1994; Hanski and Ovaskainen 2002). Native biodiversity is therefore under threat as urban areas continue to expand and replace natural habitats, yet the processes enabling wildlife to persist in urban areas are not well understood. Consequently, urban planning and management decisions often fail to ensure the long-term conservation of urban biodiversity.

To effectively address this challenge requires an understanding of how anthropogenic landscape change impacts urban wildlife, and how various species respond to landscape change and the intensification of disturbance pressures on remnant habitats. At the landscape-level, the impacts of urban development include the loss and fragmentation of natural habitats, loss of landscape connectivity and the modification of the internal composition and structure of remaining ecosystems (Forman 1995; Baskin 1998; Wilcove et al. 1998; Marzluff and Ewing 2001; Faulkner 2004). These changes in landscape composition, configuration, and ecosystem integrity have important consequences for the persistence of native fauna populations. In addition to the primary effects of habitat loss and fragmentation (Fahrig 1997, 2001), additional associated pressures include: increased hostility of the urban matrix including barriers (e.g., roads) to species dispersal and pet predation; increased edge effects; and, increased isolation of suitable habitats (Knutson et al. 1999; Villard et al. 1999; Gibb and Hochuli 2002; Wood and Pullin 2002; Baker et al. 2003; Krauss et al. 2003; Verbeylen et al. 2003; Yeoman and Mac Nally 2005). Additional local-level changes in the internal structure and composition of remnant native ecosystems negatively impacts habitat quality, which can have further substantial impacts on assemblages of native fauna (Dickman and Doncaster 1987; Rowston et al. 2002; Recher 2004; Garden et al. in press).

Understanding how species respond to multiple changes in their habitat is not straightforward as the magnitude and direction of environmental changes on wildlife populations are highly species-specific (Dickman and Doncaster 1989; Wu and Hobbs 2002; Cox et al. 2003; Tischendorf et al. 2003; Chace and Walsh 2006). In addition, the relative importance of habitat characteristics at multiple spatial scales, such as the local (< 1 ha), patch (1-100s ha), and landscape-levels (100s-1000s ha), also varies between species, and across landscape contexts (Wiens 1994; Hostetler and Holling 2000; Debinski et al. 2001; Garden et al. 2006). Successfully addressing urban conservation planning and management issues therefore requires the relative importance of multi-level environmental characteristics on native species assemblages to be determined. An important question that must be answered is: What is more important for fauna assemblage persistence: local-level characteristics, such as habitat structure and composition; patch-level attributes such as patch size and shape; or, landscape-level attributes, such as habitat loss and fragmentation? Answering this question requires testing the landscape structure hypothesis, that the structure of the whole landscape and not just the patches is a significant factor influencing species occurrence and diversity (Turner 1989; McGarigal and McComb 1995; Hokit et al. 1999; Dorner et al. 2002; McAlpine and Eyre 2002). This calls for the adoption of a patch-mosaic model rather than the traditional patch-matrix model of landscape structure (sensu Lindenmayer et al. 2003), recognising that the urban matrix is spatially heterogeneous and can perform important habitat and dispersal functions for some species (Lidicker and Peterson 1999; Fahrig 2002; Opdam et al. 2003).

This challenge of conserving multiple species within urban landscapes is further complicated by the bias in current urban ecology research towards studies that focus on a single spatial scale and/or a single species or faunal group (Garden et al. 2006). Studies conducted at a single scale have been shown to explain only part of the overall impact of habitat loss, fragmentation and degradation, producing the potential for species declines to be obscured or exaggerated (Wiens 1994; Hobbs 1999). Further, if conserving a wide diversity of taxa is the management goal, then we need to understand the habitat requirements of multiple taxa rather than a single taxon. Landscapes can not be optimised based on the requirements of only a single species or faunal group and so multi-species studies are essential (Opdam et al. 2002).

In Australia, urban development is concentrated along the coastline, especially the eastern seaboard, which are areas synonymous with high species diversity and endemism (Queensland Museum 1995). A prime example is south-east Queensland, an area of rich flora and fauna diversity and endemism (Queensland Museum 1995), and also the nation's most rapidly urbanising region, with the population predicted to increase by more than one million people over the next two decades (The State of Queensland 2006). In these regions, while there is increasing urban research effort into highly mobile, easily identifiable bird species (e.g., Catterall et al. 1989; Grover and Slater 1994; Bentley and Catterall 1997; Catterall et al. 1998; Catterall 2004), and high profile mammal species such as the koala (*Phascolarctos cinereus*) (Smith and Smith 1990; Dique et al. 2004; McAlpine et al. 2006a,b; Rhodes et al. 2006), fewer studies have considered the impact of urbanisation on reptile, amphibian and small mammal species (Garden et al. 2006). Consequently, urban land managers are often ill-informed regarding how best to conserve these taxa in the region's rapidly expanding urban footprint. The adoption of a multi-scaled, multi-species approach to urban ecology studies would provide information for successfully achieving urban wildlife conservation.

We addressed the question: What is more important for native species richness in fragmented urban landscapes – local, patch, or landscape-level influences? To answer this question, we examined native terrestrial reptile and small mammal species compositions within lowland remnant vegetation fragments of Brisbane City, south-east Queensland (Australia). The focus was on explanation rather than on predictions, although we recognise the two are complementary. We first developed a set of *a priori* models based on the conceptual model of the effects of urban-driven habitat loss, fragmentation and degradation on species richness (Figure 5.1). The level of observation was the site, with mammals and reptiles surveyed twice over two consecutive years at 51 sites. We applied the information-theoretic approach of Burnham and Anderson (2002) to the data in order to investigate how local, patch and landscape-level attributes influence species richness. Model averaging was used to account for model selection and parameter uncertainty. Using this process, the multi-level model was found to provide the best explanation of reptile and small mammal species richness within survey sites.

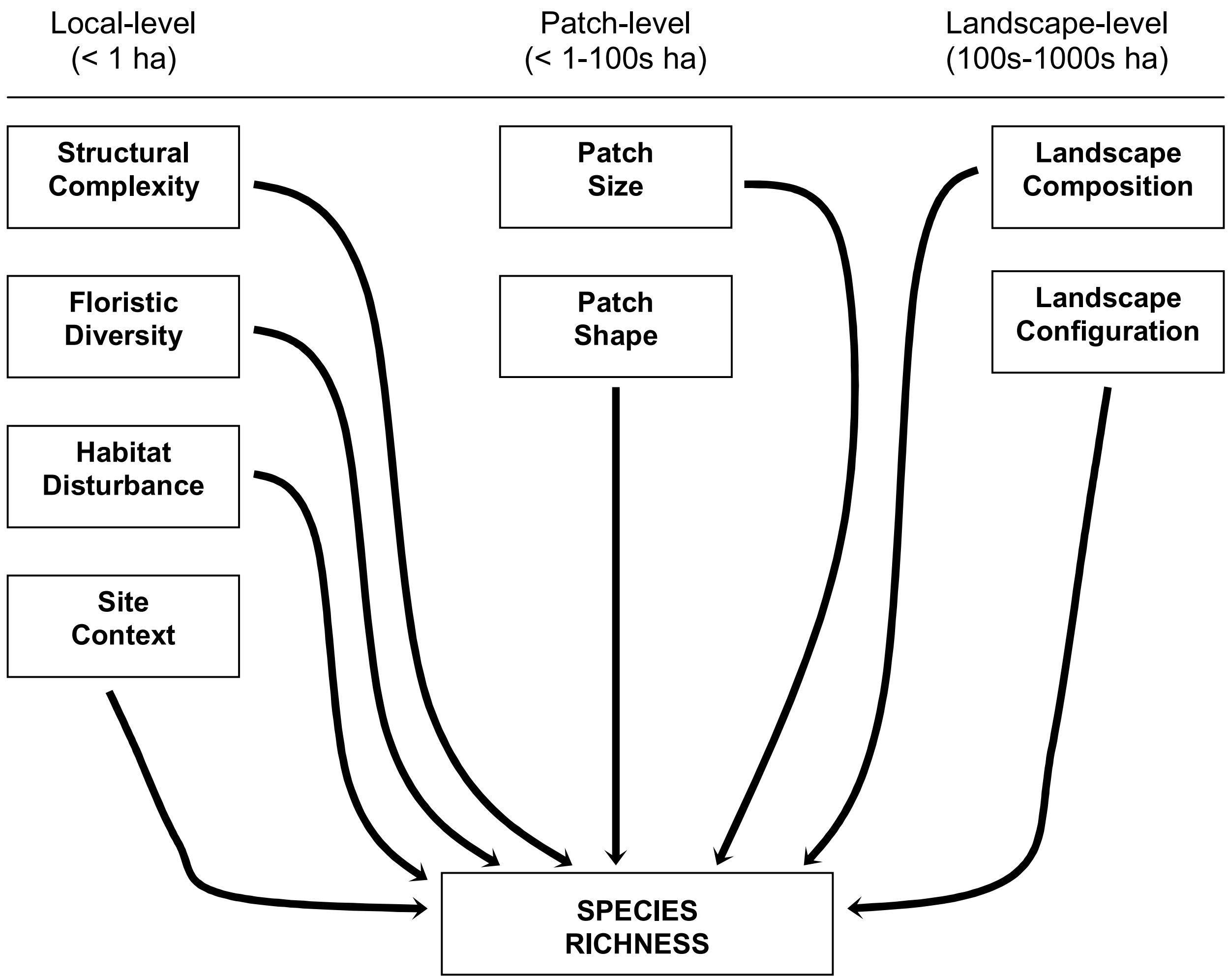


Figure 5.1. Conceptual model showing factors influencing species richness in fragmented urban landscapes. A series of *a priori* predictors are embedded in this model.

5.3. *A PRIORI* MODELS

The testing of *a priori* models is becoming increasingly common (Hilborn and Mangel 1997). Some believe that developing and testing *a priori* models is superior to null hypothesis testing in '…making inferences about observational data, especially when data are collected from complex systems…' (Johnson and Omland 2004, p.106). The following alternative *a priori* models were developed based on accumulated knowledge of landscape ecology and reptile and small mammal biology.

Model 1. "Home, sweet, home": Habitat characteristics at the local-level have the strongest influence.

This model predicts that habitat characteristics at the local-level have the highest relative influence on small mammal and reptile species richness. Key local-level habitat attributes include structural complexity (e.g., vertical vegetation structure, fallen wood, and termite mounds), floristic diversity (e.g., abundance of various plant species) and disturbances (e.g., fire, weed cover, and litter dumping). Structural complexity is expected to be more important than floristic diversity for determining species richness, with structurally complex habitats expected to support high species richness. For terrestrial reptile and small mammal species, vertical and horizontal structural elements have been shown to provide essential shelter, foraging, breeding, and thermoregulation requirements (Halliger 1993; Cork and Catling 1996; Smith et al. 1996; Burrow et al. 2001; Chambers and Dickman 2002; Jellinek et al. 2004). These structural elements are also likely to allow small animals to move at a local scale with reduced predation risk.

Model 2. "Bigger is better": Patch size and shape have the strongest influence.

This model predicts that species richness increases with patch size and shape complexity. Conceptually, this model draws from traditional theories of island biogeography (MacArthur and Wilson 1967), meta-population dynamics (Levins 1970), resource concentration (Root 1974) and edge effects (e.g., Lidicker 1999). Together, these imply that larger, more compact patches will provide more high quality habitats with a diversity of resources, and that animals are more likely to occupy and persist in these patches (Bowman et al. 2002).

Model 3. "Thinking outside the box": The area of suitable habitat has the strongest influence.

This model predicts that the richness of reptile and small mammal assemblages is primarily influenced by the area of suitable habitat in the surrounding landscape, a measure of landscape composition, and that loss of this habitat will negatively influence species richness. Fahrig (1997, 2001, 2003) argues that habitat loss, rather than habitat fragmentation, is the major driver of population declines. Several researchers nationally and internationally also support this argument. Wood and Pullin (2002), for instance, propose that butterfly species in a British landscape rely more on the availability of suitable habitat, rather than their dispersal abilities, for survival (see also: Bender et al. 1998; Carlson 2000). Within Australia, McAlpine et al. (2006b) conclude that the amount of habitat had a strong positive influence on koala occurrences in urbanised landscapes of southeast Queensland (see also: Catterall et al. 1998). We would expect, therefore, that the amount of suitable habitat would have a similar strong influence on reptile and small mammal species richness in the region

Model 4. "Location, location, location": Landscape configuration has the strongest influence.

This model predicts that the spatial configuration of suitable habitat has the strongest influence on the richness of small mammal and reptile species. The independent effects of habitat loss *per se* are often difficult to discriminate from effects of the closely related process of habitat fragmentation (Andrén 1994; Wilcove et al. 1998; Cushman and McGarigal 2002; Fuhlendorf 2002; Bender et al. 2003). Fragmentation alters the spatial arrangement of habitat patches by reducing mean patch area, altering patch shapes, and increasing both the number and isolation of patches. We conceptualise that the urban landscape is a mosaic of habitats of different quality, and the spatial configuration and connectivity of these habitats, particularly high quality remnant forest habitats, has the strongest influence on species richness.

Model 5: Roads have the strongest influence.

This model predicts that the proximity and density of roads adjacent to and surrounding suitable habitat fragments has the strongest influence on reptile and small mammal species richness. Roads are a significant element in urban areas, causing substantial habitat fragmentation as they bisect the landscape, acting as significant dispersal barriers and increasing the probability of mortality (Forman and Alexander 1998; Forman 1999; Jaarsma and Willems 2002; Forman et al. 2003). These direct impacts, in addition to a suite of indirect impacts, have been shown to significantly influence: habitat quality, individual species, species assemblages and populations. Consequently, we expect that reptile and small mammal species richness will decrease where roads are adjacent to habitat fragments, and as road density in the surrounding landscape increases.

Model 6. "Life, the universe, and everything": The global model is the best predictor.

The global model predicts that a combination of factors occurring at the site, patch and landscape-levels will determine the distribution and richness of species living in fragmented urban landscapes. This model combines the influences and predictions outlined in models 1-5. Verboom and Apeldoorn (1990) and Fuhlendorf et al. (2002) concluded that their respective multi-level models provided the best prediction of species distributions for red squirrels (*Sciurus vulgaris* L.) in the Netherlands, and North American lesser prairie-chickens (*Tympanuchus pallidicinctus*), respectively. Despite the lack of rigorous testing, the value of multi-level models have been endorsed by previous researchers who have concluded results from single-level studies could have been improved had more variables at multiple levels been considered (e.g., Mac Nally et al. 2000).

5.4. METHODS

5.4.1. Study Area

The study was conducted in lowland (< 100 ASL) remnant vegetation patches of Brisbane City, south-east Queensland (153º2'S, 27ºE) (Figure 5.2). Brisbane is the capital of Queensland and is Australia's third largest city (area 1,220 km^2, population > 1 million) (Commonwealth of Australia 2003b). Almost 70% of the original vegetation cover has been cleared since European settlement began in 1824, with lowland forest ecosystems the most heavily cleared (Catterall and Kingston 1993). Contemporary urban development is occurring in the lowland outer suburbs to the south and southeast (Brisbane City Council 2001). Despite the extensive loss of these lowland habitats, Brisbane still maintains high fauna and floristic diversity, with the highest vertebrate diversity and endemism of any Australian capital city

(Queensland Museum 1995). The current study focused on these lowland fragmented habitats in the city's southern and eastern suburbs (Figure 5.2).

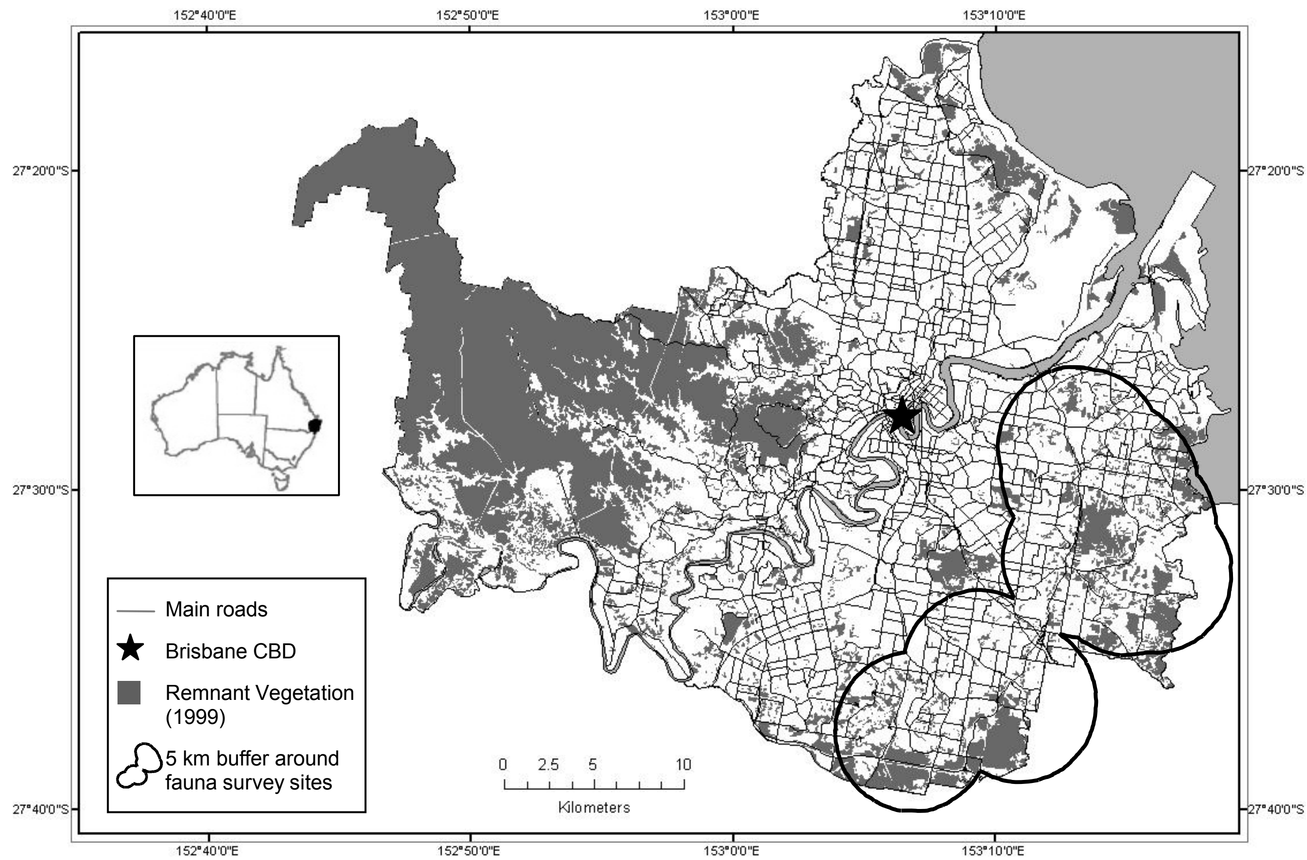


Figure 5.2. Map of Brisbane City local government area showing central business district (CBD) relative to the study area within which fauna surveys were conducted.

5.4.2. Habitat Mapping, Stratification and Site Selection

Of the 59 sites initially selected, eight were eliminated from the data set due to fire or substantial human interference preventing the full complement of surveys to be conducted. Therefore, data collected from 51 sites, which were able to be surveyed twice, were used for final analyses. Survey sites locations were initially selected using Brisbane City Council (BCC) satellite imagery and GIS data. GIS remnant regional ecosystem layers overlayed on recent satellite imagery were used to identify possible survey site locations within regional ecosystem type 12.9-10.4. This lowland ecosystem is dominated by scribbly gum (*Eucalyptus racemosa*) communities growing on sandy soils and is one of the most fragmented lowland ecosystem types in the region (Young and Dillewaard 1999). Stratification of site selection by regional ecosystem type prevented potential differences in species assemblages due to natural species variations between different ecosystems.

Potential site locations were then assessed in situ to determine their suitability. Sites were considered unsuitable if they were largely disturbed, had a high probability of human interference, or were difficult to access. To ensure independent samples of species communities, sites were situated at least 200 m apart. In remnant fragments that received high human use, sites were additionally located at least 20 m from designated walking tracks and recreational areas, to reduce the likelihood of human interference. In order to fulfil these limiting requirements, sites were necessarily located on local council and privately owned properties.

5.4.3. Wildlife Surveys

The 51 sites were surveyed twice for the occurrence of native terrestrial reptile and small mammal occurrence (Garden et al. in review a). Sites were first surveyed in spring/summer of 2004 with repeat surveys conducted in spring/summer of 2005. For each survey period, four sites were surveyed simultaneously over three consecutive nights. A combination of live-trapping (cage, Elliot, and pit-fall traps), direct observation and trace (hair funnels, tracks, scats, and vocalistations) survey methods were employed to maximise the probability of detecting target reptile and small mammal species. The standard Australian mammal bait mixture (Menkhorst and Knight 2001) was used to bait cage traps, Elliott traps and hair funnels, whilst pit-fall traps were left unbaited. Records from both survey periods were collated into a single data set (total 306 survey nights) for the purposes of data analysis.

5.4.4. Explanatory Variables

Site-level

Site-level habitat variables were recorded for each site (Table 5.1). These included measures of structural complexity, floristic diversity and human-disturbance. Structural complexity was measured as: average percent canopy, midstorey, understorey and ground cover; midstorey density; ground cover complexity and depth; volume of fallen woody material; number of pieces of fallen woody material containing hollows; number of terrestrial termite mounds; and soil compaction. Floristic diversity was measured by counting the total number of *Acacia spp.*, *Allocasuarina spp.*, *Banksia spp.*, *Callistemon spp.*, *Melaleuca spp.*, *Pteridium esculentum*, and *Xanthorrhoea spp*. Human disturbance was measured as the presence of waste litter, sawn wood, fire scars, and average percent weed cover at each site.

Table 5.1. Key explanatory variables, as determined following initial multi-scaled, multivariate exploratory analyses, that were used in the analysis of reptile and mammal species richness. For each variable, the spatial level at which it was measured, the environmental characteristic it measures, and a brief description are shown.

Spatial Level	Environmental Characteristic	Explanatory Variable	Description
Local (< 1 ha)	Floristic diversity	Grass trees Paperbarks	Count of total number of Banksias (*Banksia spp.*), grass trees (*Xanthorrhoea spp.*) or paperbarks (*Melaleuca spp.*) at each site.
	Disturbance	Weed cover	Presence or absence of weeds at each site.
	Structural complexity	Wood volume (m^3)	Volume of fallen woody material at each site.
		Soil compaction	Average measure of soil hardness at each site measured as: number hits (≤ 20) required to drive a weighted soil probe 20 mm into ground – 30 measures per site.
Patch (1-10 ha)	Size	Patch size (ha)	Area of forest patch within which each site was located.
	Shape	Patch shape	Measure of the complexity of forest patch shape.
Landscape (10- 100s ha)	Composition	Proportion of landscape occupied by each habitat class (%)	Percentage of the landscape occupied by each class type: forest, rural, built, and road reserve, measured at each spatial extent.
		Shannon's evenness index	Measure of evenness of all habitat classes in the landscape.
	Configuration	Mean patch area (ha)	Average area of patches for each habitat class measured at each spatial extent.
		Patch mean nearest neighbour (m)	Average minimum distance between patches of each habitat class measured at each spatial extent.
		Patch connectivity (%)	Average percentage of functional joins between patches of each habitat class measured at each spatial extent.
		Landscape cohesion index	Measure of physical connectedness between patches of each habitat class measured at each spatial extent.
		Interspersion and Juxtaposition Index	Measures spatial arrangement and "mixing" of all habitat classes in the landscape.
		Landscape contagion	Measure of spatial dominance of one habitat class in the landscape.

Other site-level measurements included distance from each site to the closest: patch edge, permanent water source, sealed road, unsealed track, cleared area, and built structure. These measurements were derived from Quickbird high resolution satellite imagery (resolution = 2.5 m). Land tenure was recorded for each site as certain habitat patches span tenure boundaries and so local-level management may vary within patches, thereby influencing species occurrence.

Patch-level

Patch-level measurements included the patch size (ha), perimeter length (m), and shape (relative to a circle of equal area). All patch-level variables were calculated using ArcGIS version 9.1 (ESRI Inc. 1999-2005) derived from BCC land use data and Quickbird high resolution satellite imagery.

Landscape-level

Quickbird imagery and BCC land use data were used to classify the landscape into three habitat classes: forest, rural and urban (Table 5.2) which were converted to a raster layer with a 3 m pixel size. Landscape structure (composition and configuration) was then quantified at six radial extents (500 m, 1 km, 2 km, 3 km, 4 km and 5km) around each site using FRAGSTATS Version 3 (McGarigal et al. 2004). Landscape composition was measured as the proportion of landscape occupied by each habitat class and the dominance of one particular habitat class, as measured by Shannon's Evenness Index. Roads were incorporated in the built habitat class for measurements of the proportion of landscape occupied by urban. However, previous research has highlighted the significant negative influence that roads can have on fauna species and so, the effect of road density was examined independently by measuring the proportion of landscape occupied by road reserves in each spatial extent. Landscape configuration was measured using forest patch and edge density, forest mean patch area and shape complexity, Euclidean mean nearest neighbour distance between forest patches, and the connectivity of forest patches.

Table 5.2. Habitat classes and their perceived function for reptile and mammal species. Although roads are encompassed within the 'built' habitat class, their independent influence on species richness has also been investigated; a description of roads is therefore also presented in italics.

Habitat Class	Characteristics	Function
Forest	• Low-density human use dominated by remnant native vegetation; Remnant was defined as the canopy, midstorey, understorey, and ground layers being >70% intact.	High-quality foraging and breeding habitat with low-risk movement.
Rural	• Moderate-density human use dominated by cleared/open grassy areas with scattered trees and parklands. • Includes: agriculture and grazing paddocks; managed parklands, residential acreage, open sports ovals, and golf courses.	Low-moderate quality habitat with moderate-risk movement, depending on species behavioural attributes.
Built	• High-density human use dominated by man-made structures. • Includes: residential and industrial estates, mining quarries, commercial, business, education and sporting centres, car-parks, and road and railway networks.	Unsuitable habitat with high-risk movement and high frequency of barriers.
Road Reserve	• *Man-made area encompassing sealed road pavements and associated road verges.* • *Variable in width and dominated by low-high vehicle use.*	*Subdivides habitats and presents significant, high-risk barriers to movement*

5.4.5. Statistical Analyses

Exploratory data analysis

The response variable was the total number of target species detected for each site for both survey periods. Mammals and reptiles were analysed independently. Exploratory data analysis was conducted in the following series of steps. First, explanatory variables were standardised to have a mean of 0 and a standard deviation of 1 to allow comparison of model parameter estimates. Then, univariate generalised linear modelling using the Gaussian family within R version 2.0.1 (The R Development Core Team 2004) was applied to model species richness against each explanatory variable. The univariate models were then ranked according to their significance and Akaike's Information Criterion (AIC) (Akaike 1983) of each model. We also tested for statistically important interaction effects between forest amount and forest patch density, and between forest amount and forest mean nearest neighbour distance. AIC was calculated as:

$$\mathrm{AIC} = -2L + 2K \qquad (1)$$

where L is the model's log-likelihood and K, the number of parameters in the model.

Second, for forest amount, forest patch and edge density, forest mean patch area, forest mean nearest neighbour distance, and road density, we chose the spatial extent with the highest Akaike weight value (Akaike 1983; Burnham and Anderson 2002). The Akaike weight (w_i) of a model is the relative likelihood of the model compared to all other models in the set with weight values > 0.8, indicating a high level of support for a candidate model (Burnham and Anderson 2002). Next, to test how the effect of forest amount, configuration and road density changed with the spatial extent of analysis, we plotted the Akaike weight against the metric mean and standard deviation for each spatial extent, and examined if the Akaike Weight followed a similar pattern to the mean and standard deviation.

Finally, we tested for co-linearity among this subset of explanatory variables using Spearman's Rank correlation. A correlation coefficient of ≥ 0.7 was chosen to identify pairs of highly correlated variables. The decision to retain or remove variables was based on its perceived ecological significance for determining species richness as well as its level of co-linearity. Following these exploratory processes, the set of explanatory variables was reduced to a subset of 11 key variables for both reptiles and mammals.

Model averaging

Multivariate generalised linear models were then used to model species richness against the subset of key explanatory variables for each taxa. Because of the high uncertainty in the final model selection, we applied a model averaging approach using R (Burnham and Anderson 2002). Model averaging constructs alternative models from all possible linear combinations of the subset of explanatory variables and interactions. The alternative models were then ranked by their AIC value and Akaike weight (w_i). We assessed model uncertainty by successively summing the Akaike weights of the top models until the cumulative sum of weights $\geq$ 0.95.

Because model uncertainty was high for both reptiles and mammals, we calculated the model-averaged parameter estimate by weighting the parameter estimate against the Akaike weight for each model. We also calculated the unconditional standard error of each parameter estimate according to Burnham and Anderson (2002, p. 162). Finally, we ranked the variables according to their relative importance by summing the Akaike weight (Σw_i) from all model combinations where the variable occurred. The larger the sum of the weight value, the more important the variable is relative to the other variables.

The independent effect of key explanatory variables and interaction terms was examined using the hierarchical partitioning package, "hier.part", in R 2.0.1 (Walsh and Mac Nally 2005). Hierarchical partitioning enables the independent and joint percent contribution of each variable to the overall explanatory power of the model to be determined (Chevan and Sutherland 1991; Mac Nally 2000). Variables that are highly collinear have a joint effect larger than their independent effects.

Model fit

To asses the fit of the final models we plotted the observed and predicted species richness for mammals and reptiles. We applied an arbitrary threshold of error of $\pm$ 1 species, and calculated the number of sites within this range. Spatial autocorrelation in the Pearson's residuals were investigated using the ROOKCASE add-in in Microsoft Excel (Sawada 1999) to calculate Moran's index and construct correlograms. The correlograms were compared to those derived from the observed species richness response variables for reptiles and mammals. Spatial autocorrelation was considered statistically significant if Znormal < 1.96.

5.5. RESULTS

A total of 19 reptile species (eight families) and nine mammal species (three families) were detected overall. Observed species richness varied between zero to five species for reptiles, and zero to three species for mammals.

5.5.1. Spatial Extents of Landscape-Influence

For reptiles, the univariate generalised linear models identified four site, three patch and 15 landscape-level variables as being significant ($p < 0.10$ as identified by R). There was no single spatial extent at which key landscape-level variables had the strongest effect (Figure 5.3). The proportion of the landscape occupied by forest had the strongest support at the 1 km extent ($w_i = 0.347$) and little support at 3-4 km extents (Figure 5.3a). For forest mean patch area, there was weak support for the 4 km spatial extent (Figure 5.3b) relative to the other extents, while for forest patch connectivity there was weak support at the 5 km spatial extent (Figure 5.3d). In contrast, the proportion of landscape occupied by roads had the strongest level of support at the 500 m extent ($w_i = 0.419$; Figure 5.3c). For mammal species richness the proportion of the landscape occupied by forest and rural habitats combined was the most important of the landscape-level variables. This variable had the strongest support at the 5 km extent (w_i = 0.466; Figure 5.3e).

5.5.2. Subset of Explanatory Variables

The top 11 explanatory variables for reptiles (Table 5.3a) and mammals (Table 5.3b) showed low to moderate co-linearity. The highest level of co-linearity for variables explaining reptile species richness was between the proportion of forest occupying the landscape at the 1 km extent and forest patch size (r = 0.90). Patch size was highly correlated with forest mean patch area at 4 km (r = 0.74), forest patch cohesion at 2 km (r = 0.81), the proportion of landscape occupied by roads at the 500 m spatial extent (r = -0.76), and landscape contagion at the 1 km extent (r = 0.80). Landscape contagion at 1 km was also highly correlated with the proportion of landscape occupied by forest at the 1 km extent (r = 0.79), and with forest patch cohesion at the 2 km spatial extent (r = 0.73). Although not correlated with other explanatory variables, wood volume and patch shape were removed as they had low correlations with reptile species richness, and subsequent statistical analyses were best performed on a smaller subset of variables ($\leq$ 6). After accounting for co-linearity, the final subset of explanatory variables were: weed cover, soil compaction, proportion of the landscape occupied by forest at 1 km, forest mean patch area at 4 km, forest patch connectivity at 5 km, proportion of the landscape occupied by roads at 500 m, landscape contagion at 1 km, and the interaction between the proportion of landscape occupied by forest at 1 km and forest connectivity.

For mammals, the highest level of co-linearity was between the combined proportions of forest and rural habitats at the 5 km landscape extent and the proportion of the landscape occupied by roads at the 4 km extent (r = -0.95). The final subset of explanatory variables included: grass trees

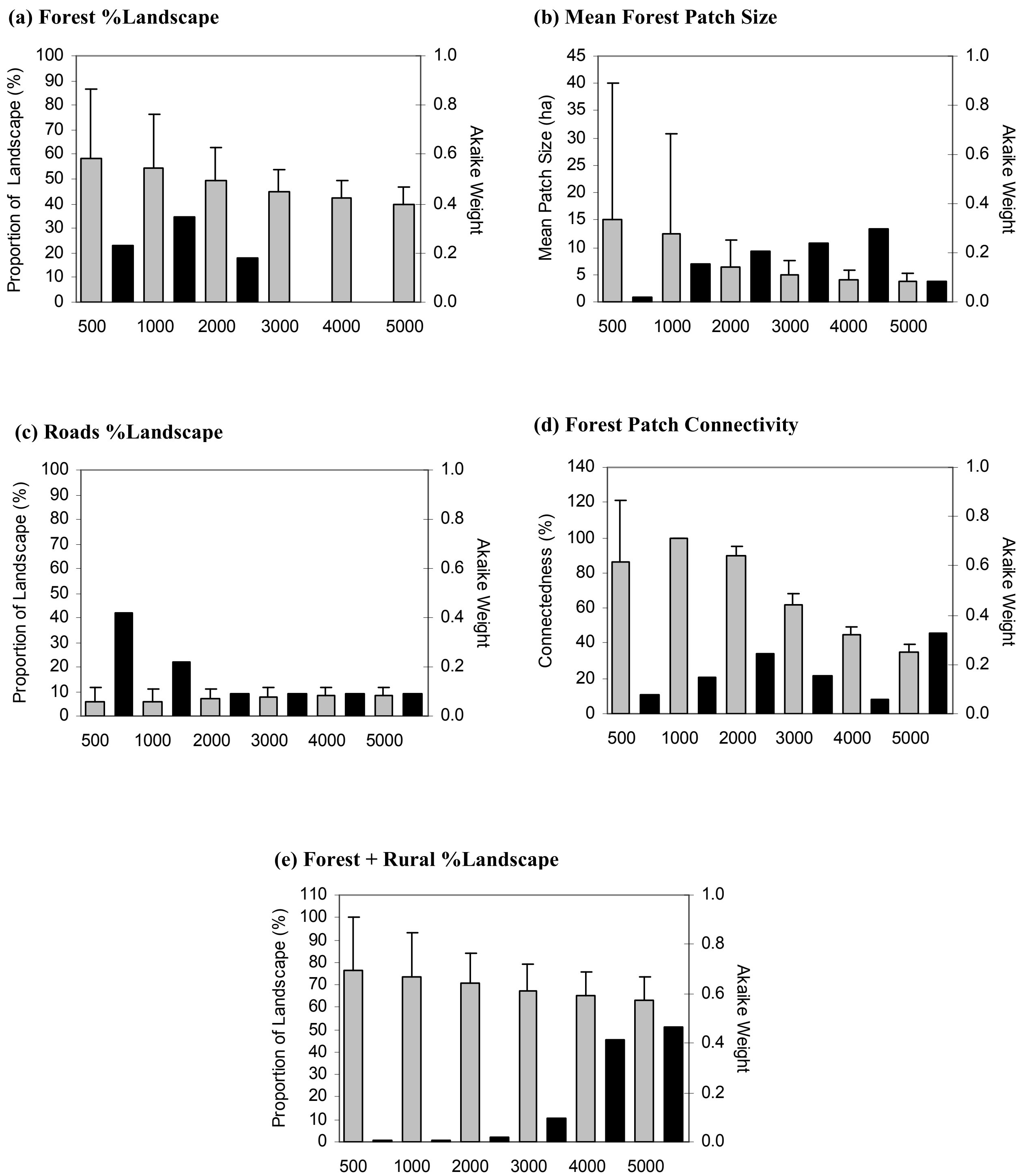


Figure 5.3. Key landscape-level predictors from the final model at increasing spatial extents (500-5000 m) from survey sites for: (a-d) reptile species richness, and (e) mammal species richness. Variable means (□), standard deviations (error bars), and Akaike weights (w_i) (■) are shown.

Table 5.3. Correlation matrix of Spearman's rank correlation coefficients for key explanatory variables for: (a) reptile species richness, and (b) mammal species richness. Correlated variables ($\geq$ 0.70) are in bold.

(a)

Variable	Var1	Var2	Var3	Var4	Var5	Var6	Var7	Var8	Var9	Var10	Var11	Var12
1. Reptile richness	1											
2. Weed cover	0.26	1										
3. Wood volume	0.13	-0.04	1									
4. Soil compaction	-0.25	0.01	-0.25	1								
5. Patch size	0.26	0.58	-0.10	0.00	1							
6. Patch shape	-0.07	0.26	-0.18	0.01	0.49	1						
7. Forest %Landscape (1 km)	0.30	0.54	-0.10	0.07	**0.90**	0.37	1					
8. Forest mean patch area (4 km)	0.36	0.42	-0.01	0.01	**0.74**	0.25	0.66	1				
9. Forest patch cohesion (2 km)	0.34	0.47	-0.09	0.04	**0.81**	0.38	**0.87**	0.66	1			
10. Forest patch connectivity (5 km)	0.32	0.19	0.32	-0.36	0.29	-0.25	0.30	0.34	0.33	1		
11. Roads %Landscape (500 m)	-0.28	-0.62	0.24	0.04	**-0.76**	-0.54	-0.69	-0.63	-0.62	-0.11	1	
12. Landscape contagion (1 km)	0.46	0.56	0.11	-0.03	**0.80**	0.24	**0.79**	0.60	**0.73**	0.42	-0.55	1

(b)

Variable	Var1	Var2	Var3	Var4	Var5	Var6	Var7	Var8	Var9	Var10	Var11	Var12
1. Mammal richness	1											
2. Paperbarks	0.14	1										
3. Grass trees	0.30	0.02	1									
4. Patch size	0.10	-0.16	-0.10	1								
5. Patch shape	0.23	0.00	0.11	0.49	1							
6. Forest %Landscape (4 km)	0.38	-0.02	0.28	0.51	0.46	1						
7. Forest + rural %Landscape (5 km)	0.46	0.22	0.36	0.03	0.27	**0.74**	1					
8. Roads %Landscape (4 km)	-0.44	-0.16	-0.36	-0.17	-0.41	**-0.83**	**-0.95**	1				
9. Forest patch mean nearest neighbour (5 km)	-0.22	-0.23	-0.27	0.37	-0.12	-0.20	-0.66	0.52	1			
10. Forest cohesion (5 km)	0.37	-0.09	0.30	0.43	0.28	**0.86**	**0.79**	**-0.80**	-0.21	1		
11. Shannon's evenness index	0.21	0.15	0.19	-0.22	0.29	0.34	**0.70**	-0.69	**-0.76**	0.27	1	
12. Interspersion & juxtaposition	-0.34	-0.19	-0.36	-0.28	-0.34	**-0.72**	-0.48	0.51	0.24	-0.52	-0.09	1

(*Xanthorrhoea spp*.) and paperbarks (*Melaleuca spp*.) at the site-level; forest patch size and forest patch shape at the patch-level; and the proportion of forest and rural habitats combined at the 5 km landscape extent.

5.5.3. Effect of Explanatory Variables

For reptiles and mammals, there was moderate model uncertainty, and a low–moderate level of uncertainty for the average parameter estimates (Figure 5.4). For reptiles, soil compaction had the strongest negative effect (Figure 5.4a). Forest mean patch area at the 4 km extent had a strong positive influence on reptile species richness, with the proportion of forest habitat at the 1 km landscape extent having a moderate positive influence (Figure 5.4a). The absence of weed cover at the site-level and forest connectivity at the 5 km extent both had a weak positive influence on reptile species richness, whilst the proportion of roads at the 500 m extent had a weak negative influence, as did the interaction between the proportion of forest patch connectivity and the proportion of forest habitat in the landscape (Figure 5.4a).

Mammal species richness was strongly positively influenced by the proportion of forest and rural habitats combined at the 5 km landscape extent (Figure 5.4b). The presence of paperbarks and grass trees at the site-level both had an important influence, whilst at the patch-level, patch size and shape had weak positive effects on mammal species richness (Figure 5.4b).

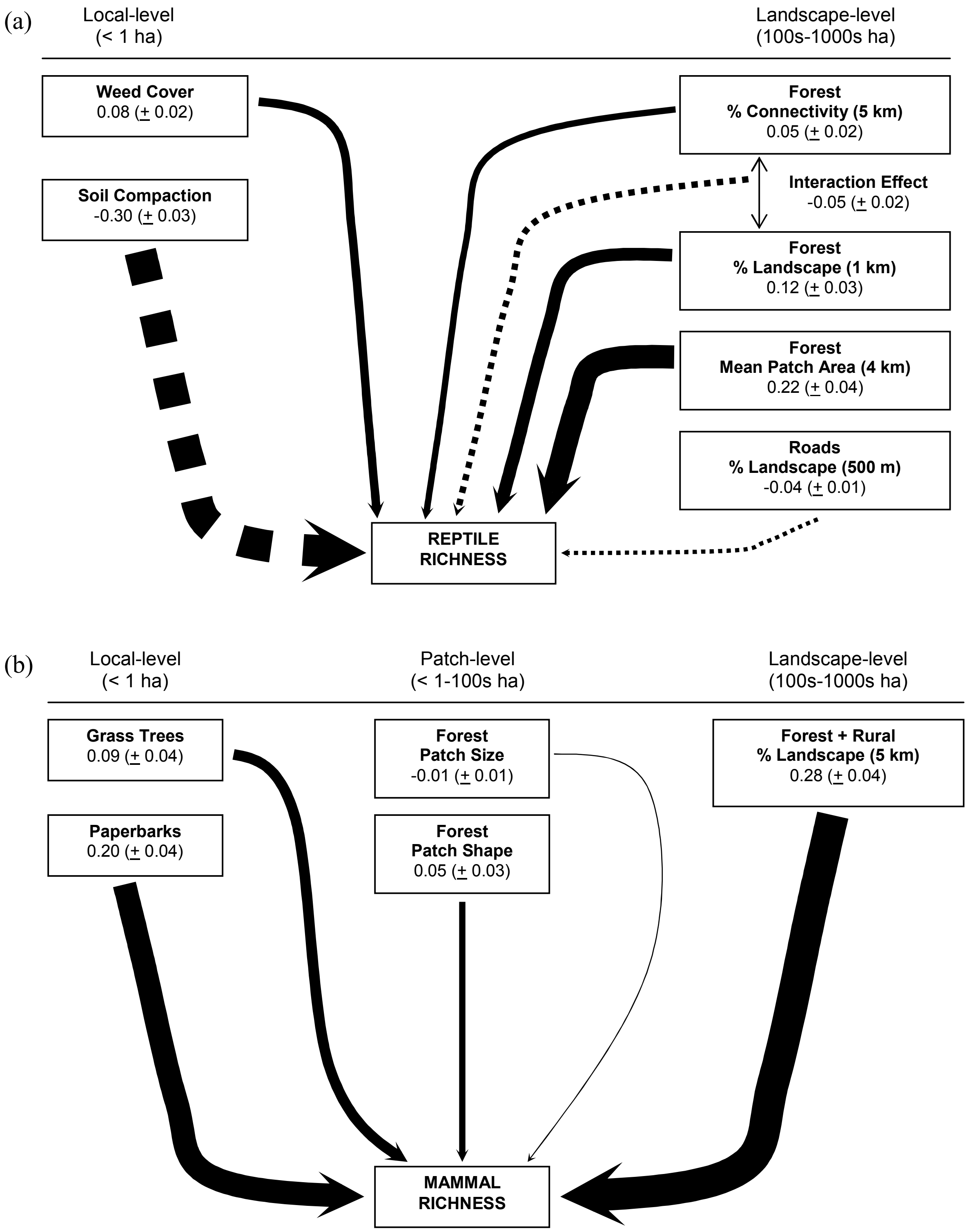


Figure 5.4. Path diagrams showing average parameter estimates and unconditional standard errors of the estimates for each explanatory variable present in all model combinations for: (a) reptiles, and (b) mammals. Line widths are proportional to the average parameter estimates. Negative effects are indicated by dashed lines.

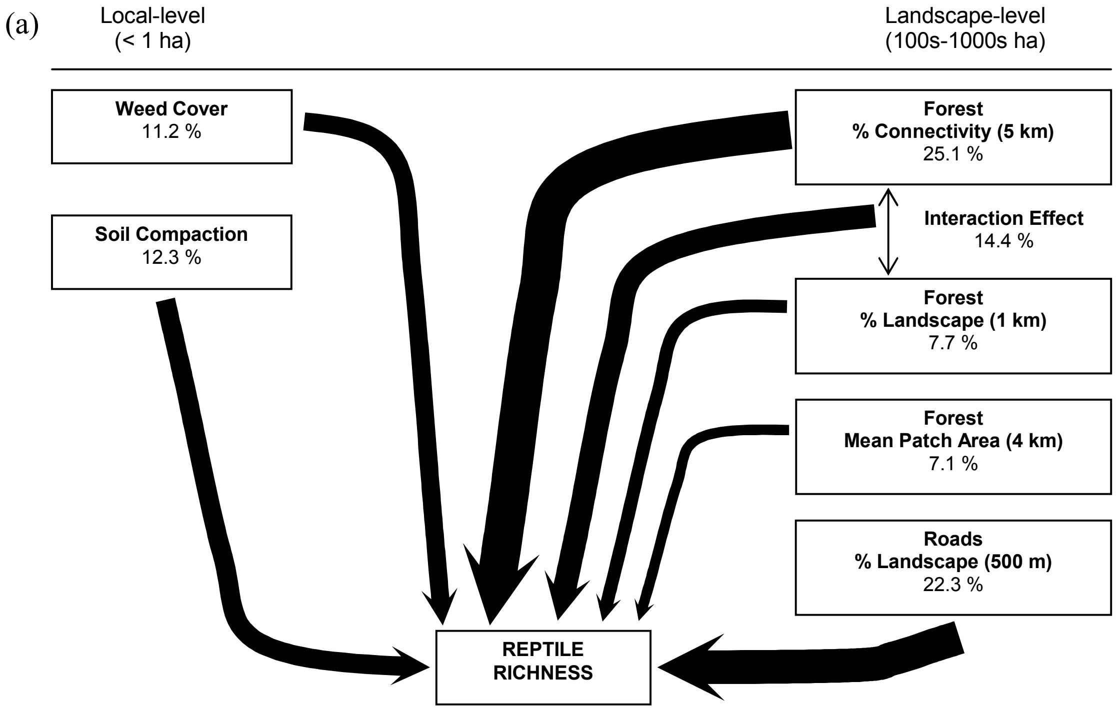
5.5.4. Independent Effects and Ranking of Explanatory Variables

The best-fit multivariate generalised linear model for reptiles explained 25.6% of the observed variation in reptile species richness (Figure 5.5a). For all variables, the independent effect was greater than the joint effect. Almost 50% of the explained variation was due to forest connectivity at 5 km and the proportion of roads at the 500 m landscape extent. The interaction effect between forest connectivity and the proportion of forest in the landscape also had a strong independent effect. The proportion of forest habitat in the landscape at 1 km and the forest mean patch area had weaker independent effects. At the site-level, soil compaction and the absence of weeds had similar independent effects.

For mammal species richness, the final model containing all explanatory variables explained 27.1% of the observed variation (Figure 5.5b). The proportion of forest and rural habitat combined contributed almost 40% of the overall variation in mammal species richness. The abundance of paperbarks and grass trees explained 28.7% and 19.5%, respectively, while forest patch shape and patch size explained 9.8% and 2.9%, respectively.

The ranking of the key explanatory variables according to the sum of the Akaike weights showed a somewhat different pattern (Figure 5.6). Hierarchical partitioning teases apart the independent effect of each variable, while the sum of the Akaike weights provides a measure of the relative importance of each explanatory variable for explaining the response variable (McAlpine et al., 2006b). For reptiles, soil compaction had the highest rank ($w_i = 0.768$), followed by forest mean patch area and the proportion of landscape occupied by forest habitat at the 1 km extent (Figure 5.6a). Weed cover, forest connectivity, the interaction between forest connectivity and proportion of forest in the landscape, and the proportion of the landscape occupied by roads had lower rankings.

The ranking of explanatory variables for mammal species richness according to the sum of the Akaike weights (Figure 5.6b) showed a similar pattern to the independent effect derived from the hierarchical partitioning analysis (Figure 5.5b). The combined proportions of forest and rural habitats in the landscape had the highest Akaike weight value ($w_i = 0.869$) (Figure 5.6b). Paperbarks were ranked next followed by grass trees, patch shape and patch size respectively.



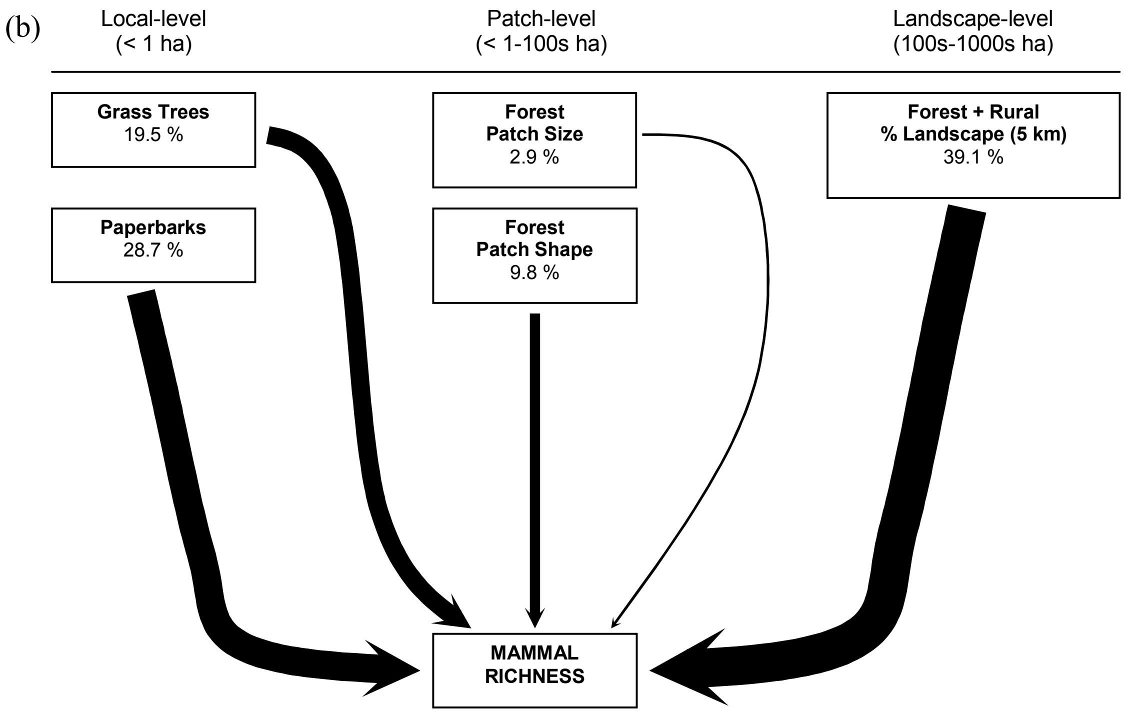


Figure 5.5. Path diagrams showing independent effect of each explanatory variable as a percentage of overall percent explained in all model combinations of key explanatory variables. Line widths are proportional to the independent contribution of each variable.

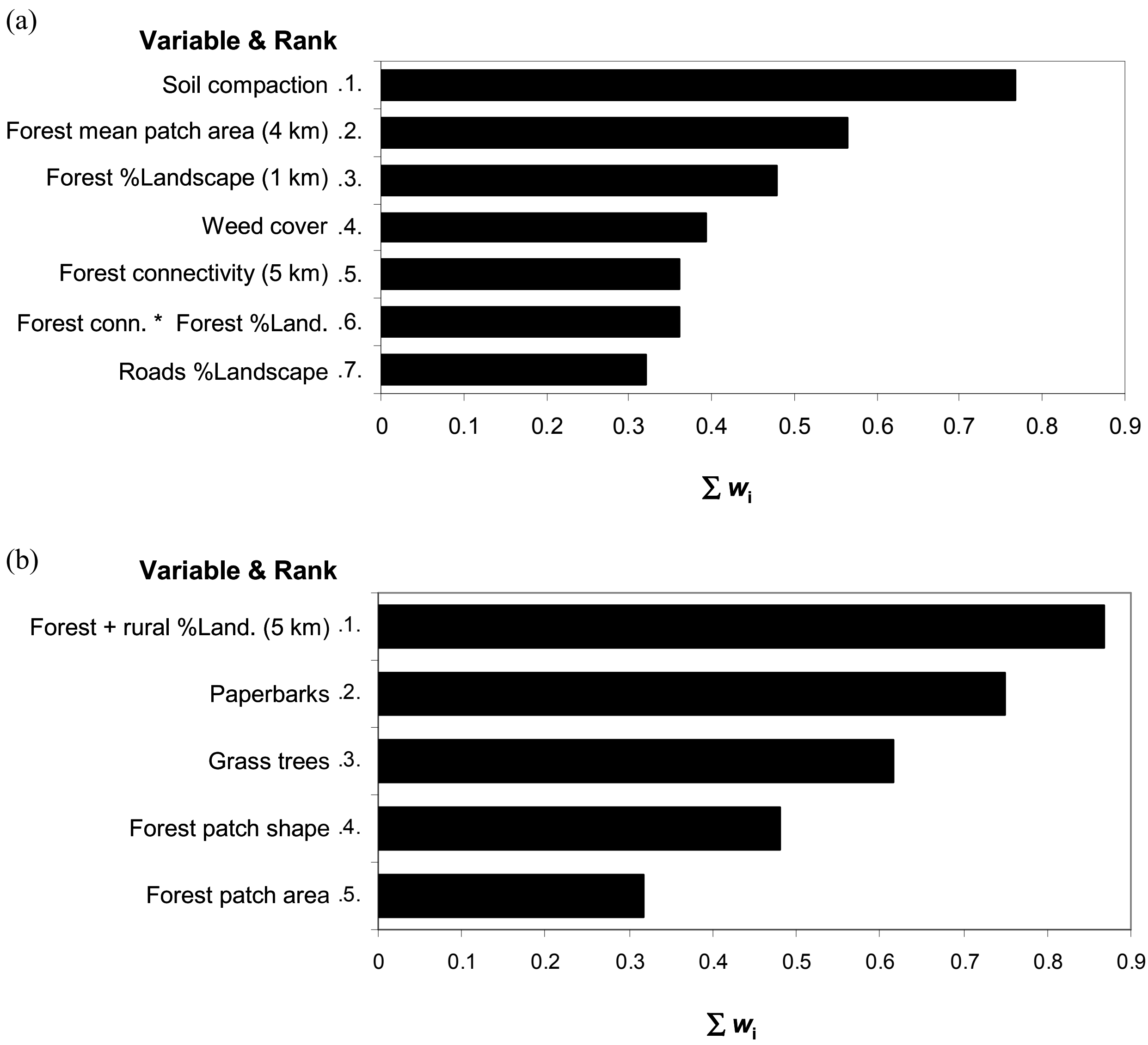


Figure 5.6. Relative importance of key explanatory variables for (a) reptile species richness, and (b) mammal species richness. Variables are ranked according to the sum of the Akaike weights ($\sum w_i$) for each variable.

5.5.5. Spatial Autocorrelation and Model Fit

There was weak spatial autocorrelation in the residual Moran's index values for reptiles and mammals, with these values not being statistically significant (Znormal < 1.96). The mapped distribution of the model residuals for reptiles and mammals showed relatively even distributions (Figure 5.7). For reptiles, the residual distribution was relatively even, although there was some clustering of northern sites where the highest deviation from the observed species richness occurred (Figure 5.7a). The best fitting model for reptiles correctly predicted ($\pm$1 species) the observed reptile species richness for 43 out of the 51 fauna survey sites. At 12 sites, the predicted species richness was equal to the observed species richness. Six sites were predicted to be within $\pm$2 species and at one site, three more species were observed than was predicted.

For mammals, the greatest deviation between observed and predicted species richness occurred in northern sites (Figure 5.7b). The best fitting model for mammal species richness had a higher accuracy than the model for predicting reptile species richness. Of the 51 survey sites, mammal species richness predicted 47 sites within $\pm$1 species, with 60% of these sites correctly predicted (i.e. no error). Three out of the 51 sites were predicted to have two species more than were observed and one site was predicted to have three less species than were observed.

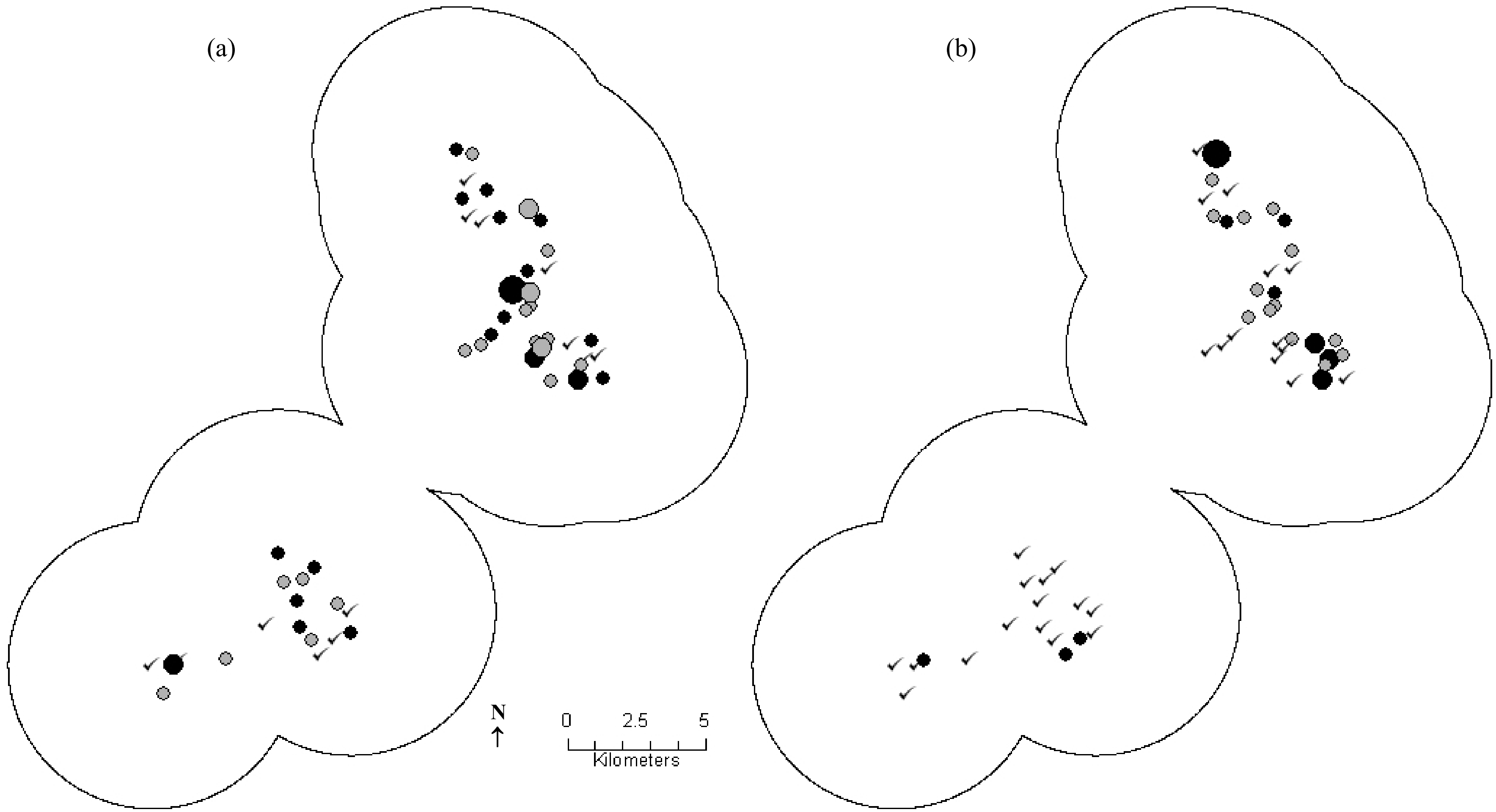


Figure 5.7. Distribution of model residuals for: (a) reptile species richness, and (b) mammal species richness. Dot size indicates magnitude of residual error: ○ 1 species; ◯ 2 species; ◯ 3 species. Black dots: observed > predicted; Grey dots: observed < predicted; Ticks: observed = predicted. Black line surrounding symbols indicates 5 km extent from all fauna survey sites.

5.6. DISCUSSION

This study showed that terrestrial reptiles and small mammals in Brisbane's lowland, remnant fragments were primarily influenced by landscape composition and configuration, followed by local-level habitat quality. Patch shape and size were important only for mammal species. These findings demonstrate the need to explicitly include the landscape context as part of multi-level investigations of factors influencing species-habitat relationships in fragmented urban landscapes. The study also highlights differences in key habitat influences among taxa. Understanding these differences as well as their relative importance for different taxa is essential if researchers are to adequately inform urban land managers and planners of best-management actions for effective and efficient urban biodiversity conservation practices.

The outcomes from this study are consistent with previous studies that have also posited the combined importance of variables across multiple spatial scales. For example, a multi-scale study by Fuhlendorf et al. (2002) on lesser prairie-chicken (*Tympanuchus pallidicinctus*) populations in the southern Great Plains of North America concluded that an "…individual spatial scale…would not have given completely accurate results…" (p. 626). Verboom and Apeldoorn (1990) also found that attributes at the local, patch and landscape-levels provided the best predictive power for determining the occurrence of red squirrels (*Sciurus vulgaris*) in fragmented forests of The Netherlands, while Fischer et al. (2004) found that reptiles in fragmented grazing landscapes of New South Wales, Australia, responded to habitat variables across multiple spatial scales. Like our study, these previous studies also show that the importance of specific habitat attributes at each level may vary between species.

Reptiles

Reptile species richness was influenced by attributes at both the local and landscape-levels. High reptile richness was positively correlated with soft soils and an absence of weeds at the local-level. Hadden and Westbrooke (1996) and Jellinek et al. (2004) similarly reported that increased weediness at sites resulted in lower reptile species richness. Jellinek et al. (2004) postulated that the proportion of weed cover at a site may be indicative of the time a patch had been isolated. We did not include the time since isolation in this study due to a lack of reliable historical data for all habitat fragments. However, we recognise that the probability of different species persisting in fragments is likely to decrease with time, and that there can be a long relaxation period between isolation and species loss (Tilman et al. 1994; Possingham and Field 2001; White and Burgin 2004; Tait et al. 2005).

Soil compaction and weediness are linked to human habitat disturbances that degrade the fine-scale habitat structure and composition (Amrein et al. 2005; Bassett et al. 2005). This has important implications for the availability of suitable foraging, thermoregulation and shelter sites for reptiles (Murray and Schramm 1987; Wikramanayake and Dryden 1993; Smith et al. 1996; Vitt et al. 1998; Burrow et al. 2001; Fischer et al. 2003). Soil compaction is also likely to directly influence the occurrence of fossorial reptiles such as Storr's rainbow skink (*Calyptotis scutirostrum*) and Verreaux's skink (*Anomalopus verreauxii*) which require soft soils for burrowing.

The importance of habitat loss versus fragmentation is an important issue in landscape ecology and conservation biology. Pure habitat loss has implications only for the amount of habitat, whereas habitat fragmentation has implications only for its spatial configuration. Fahrig (2003) stresses that habitat loss has a clear negative effect on biodiversity, while the effect of fragmentation is more varied and complex, and that the negative effect of habitat loss is unlikely to be mitigated by the spatial configuration of that habitat. We found that, for reptile species richness, both the amount of forest habitat and its spatial configuration were important, with forest mean patch area having a higher relative importance than the amount of forest habitat.

The connectivity of forest habitats and its interaction with the amount of forest were also important. This is consistent with previous findings by Villard et al. (1999) and Westphal et al. (2003) for birds, Ferreras (2001) for Iberian lynx populations, and McAlpine et al. (2006a) for koalas. Our findings support the importance of large areas of well connected, high quality patches; a well recognised principle in landscape ecology and conservation biology (Bennett 1990; Bowne et al. 1999; Pirnat 2000; Castellón and Sieving 2006; Dixon et al. 2006).

The proportion of landscape occupied by roads at the 500 m spatial extent had strongest negative effect on reptile species richness. Road networks are one of the most prominent features of the urban landscape, with road surfaces and road density increasing with the density of human settlement (Forman and Alexander 1998). A common theme among studies investigating the effect of roads on native wildlife is the direct and/or indirect destructive and detrimental consequences of roads (Fahrig et al. 1995; Forman and Alexander 1998; Jones 2000; Bissonette 2002; Ramp et al. 2006). Roads subdivide forest habitats, increase forest patch isolation, and create significant barriers to the dispersal of terrestrial fauna (Andrews 1990; Forman, 1999). These results indicate that locating roads in close proximity to forest fragments will negatively affect reptile species richness within those habitats, and hence should be avoided wherever possible.

For reptiles, the results indicate that conserving high species richness in urban landscapes will best be achieved by maintaining large areas of high quality forest habitats in large patches with high connectivity between patches and low road density. It is important that local-level management of remnant fragments also aims to mitigate disturbance pressures that degrade local-level habitat quality, particularly those that increase soil compaction and weediness. The size and shape of forest fragments was not found to be important for reptiles, indicating the potential conservation value of both small and large fragments for supporting terrestrial reptile assemblages, so long as their internal condition and configuration in the landscape are managed appropriately.

Mammals

Mammal species richness was influenced by attributes at the local, patch and landscape-levels. Our study demonstrates that in order to conserve high mammal species richness in urban forest fragments it is necessary to adopt a hierarchical approach to research and management. High quality, large and irregular patches are important, but most important, is the amount of forest and rural habitat within the surrounding landscape.

The most influential parameter for determining mammal species richness at the landscape-level in Brisbane's fragmented landscapes was the combined proportion of forest and rural habitats in the landscape at the 5 km extent from fauna survey sites. This indicates that, like reptile species richness, the landscape context is important for mammal species richness.

When examined separately, the amount of forest habitat was more important than the amount of rural habitat for mammal richness. However, the combined influence was greater than either of the independent influences. Our findings, therefore, suggest that suitable habitat for certain small mammal species may encompass rural areas as well as forest areas.

The northern brown bandicoot (*Isoodon macrourus*) was the most commonly captured native mammal species. Bandicoots (Family: Peramelidae) are known to require a mosaic of open and closed vegetation habitats in close proximity in order to support their foraging and nesting behaviours (Dufty 1994; Scott et al. 1999; Chambers and Dickman 2002). The patchy mosaic of moderately open and closed habitats in peri-urban areas may be the reason for the abundance of this species in outer urban landscapes. It is reasonable, therefore, to assume that our results for mammals were largely driven by the high occurrence of bandicoots relative to other mammals, thereby increasing the relative importance of rural habitats. The other mammal families (Dasyuridae and Muridae) identified during the study are considered to be predominantly interior, forest dwelling species that do not characteristically cross forest gaps where trees are absent or at low densities with an absence of understorey vegetation cover.

The positive influence of rural habitats, in addition to forest habitats, may also indicate the importance of the landscape matrix, a concept that has been supported by researchers such as Szacki (1999), Brotons et al. (2003), Riffel et al. (2003), Dunford and Freemark (2004) and McAlpine et al. (2006a). Rural and semi-urban habitats may provide a buffer between forest habitats and can help facilitate vital dispersal movements if important structural elements for providing cover (e.g., clumps of trees/vegetation, fallen woody material) are present.

Dispersal movements are essential to promote genetic exchanges and ensure viable populations (Bissonette 2002).

At the patch-level, we found that larger, more circular patches supported a higher mammal species richness. This is consistent with island biogeography theory (MacArthur and Wilson 1967) which proposes that larger patches are more likely to be occupied by species and that these species will be less vulnerable to extinction. Similar findings regarding the positive influence of large patches has been documented for various fauna species. For example, Woinarski et al. (1999a, b, 2001) examined reptile, mammal and bird populations living on continental islands off the Northern Territory coast of Australia, reporting that overall species diversity and abundance were positively correlated with island size. Similarly, McIntyre (1995) found that larger habitat patches in a fragmented American landscape supported a higher diversity of bird species than did smaller patches.

At the local-level, species richness was highest in habitats that supported a large number of paperbarks and grass trees. These factors combined explained ~50% of the total variation in mammal species richness. They are indicative of floristic diversity, but also fulfil important structural roles (Bennett 1993; Haering and Fox 1995; Monjeau et al. 1998; Vernes 2003; Monamy and Fox 2005; Garden et al. in press). Paperbark trees, which had a stronger independent effect than grass trees, occur mainly on wet run-on areas (Queensland Museum 2003), and so may be acting as a surrogate for soil moisture or other attributes associated with these habitats. Although, the proximity of fauna survey sites to permanent water sources was investigated and found not to be important.

Grass trees provide important nesting and shelter niches for small mammals (Fox 1995; Lunney 1995; Vernes and Pope 2001; Marchesan and Carthew 2004; Spencer et al. 2005). The life cycle of grass trees is also closely linked with fire and, as such, the influence of grass trees may be indicative of historical fire regimes, an aspect that has previously been found to influence the occurrence of some small mammals in Australia (Fox 1982; Claridge and Barry 2000; Fox et al. 2003; Spencer et al. 2005). Investigating the effects of time since last fire and moisture levels, as well as time since isolation, climatic conditions, and seasonality (Szacki 1999; Claridge and Barry 2000) may help to more accurately predict mammal species richness in these landscapes.

5.6.1. Approach and Limitations

The research outcomes depended on developing a hierarchical conceptual model, which integrated *a priori* predictions of factors influencing species richness in fragmented urban landscapes. The aim of the analysis was to rank the importance of an *a priori* set of explanatory variables. We used an information-theoretic approach to achieve this aim, which was found to be useful for capturing both model and parameter uncertainty, which was often less than the parameter estimate. We believe that the information-theoretic approach is superior to the traditional statistical hypothesis testing approach in situations where there are potentially a large number of ‘plausible’ models. The low precision of the lower-ranked parameter estimates simply reflects their weak effect in the model and confirms their lower ranking.

Time and financial constraints prevented long-term fauna surveys and investigations across multiple seasons. Seasonal variation in species abundance, coupled with the cryptic nature of many of the target species are likely to have resulted in a degree of false-absence records. Accordingly, the strength and accuracy of the relative importance of the key environmental variables should be considered as a guide only. The consideration of additional local-level habitat attributes and historical disturbance regimes, as well as seasonal and temporal population variations and species dispersal abilities, are likely to strengthen the integrity of the final models.

5.6.2. Implications for Conservation

Urban conservation is a difficult but necessary challenge. For urban managers, certain environmental attributes are hard to manage or change, whereas others are relatively easy to manage/change. The key to urban conservation is determining which factors are most important for fauna species and basing management decisions on these priority factors. Our findings emphasise the need for urban management to consider the composition and configuration of forest habitats within the landscape, rather than just the preservation of suitable forest patches per se. In addition, the study highlights the importance of the matrix. Although the most spatially dominant feature in fragmented urban landscapes (Forman 1995), the importance of the matrix for conservation has been largely overlooked in research and conservation management (Verbeylen et al. 2003).

Based on our findings, minimising habitat loss, fragmentation and degradation are all important for urban wildlife conservation. Priority actions for reptiles and mammals include:

investigating options for connecting remnant fragments; protecting and purchasing high quality habitat fragments, giving priority to those that will most effectively decrease patch isolation and are within rural or low density urban areas; implementing informed strategies to manage and restore habitat structural complexity within patches; instigating revegetation programs to increase forest area and connectivity; and mitigating the ecological ramifications of roads on wildlife dispersal and habitat quality. Complementary to these actions will be the active mitigation of human-use disturbances that degrade habitat quality, and also an investigation of historical land-use influences on species assemblages, such as time since fire, which were not examined in this study.

Of lesser importance for reptiles and mammals was the floristic composition of forest fragments, and the size and shape of forest fragments, but these features should not be discounted for their conservation significance for other wildlife species. Floristic composition is likely to influence structural complexity and habitat heterogeneity at the local-level, which will provide more habitats suitable for a variety of species. The size and shape of remnant fragments will also influence the internal habitat condition, as well as the total amount of habitat in the landscape. Consequently, promoting fragments that are more robust and 'circular' in shape, rather than long and narrow, or small and convoluted will be beneficial for species richness.

(a) Burton's snake-lizard (*Lialis burtonis*)
- Family: Pygopodidae
- Carnivorous, diurnal flap-footed lizard.
- Average size: SVL 290 mm
- Distribution: All states and territories, except Tasmania.
- In Brisbane: relatively common, particularly in remnant bushland and well vegetated suburban gardens.

This photo shows a Burton's snake-lizard taking refuge in undergrowth.

(b) Red-bellied black snake (*Pseudechis porphyriacus*)
- Family: Elapidae
- Diurnal, highly venomous snake.
- Average size: Total length 1.5 - 2 m.
- Distribution: Queensland, New South Wales, Victoria, South Australia.
- In Brisbane: rare, restricted to suitable remnant habitat in peri-urban landscapes.

Photo from: http://www.users.bigpond.com/neen105/Snake2.htm

(c) Common tree snake (*Dendrelaphis punctulata*)
- Family: Colubridae
- Diurnal, non-venomous snake.
- Average size: Total length 1.2 m.
- Distribution: Queensland, New South Wales, Northern Territory, Western Australia.
- In Brisbane: common throughout, including inner suburbs with lush vegetation.

Photo from: http://www.tvwc.org/HTML/green%20tree%20snake.htm

(d) Carpet python (*Morelia spilota*)
- Family: Pythonidae
- Diurnal and nocturnal, constricting snake.
- Average size: Total length 2.5 m.
- Distribution: All states and territories, except Tasmania.
- In Brisbane: relatively common throughout, particularly in bushland and well vegetated suburbs.

Photo from: http://www.pbase.com/laine82/image/51706801

Plate 6. Snakes and snake-lizard: (a) Burton's snake-lizard; (b) Red-bellied black snake; (c) Common tree snake; (d) Carpet python.

Chapter 6

CONSERVING NATIVE TERRESTRIAL REPTILES AND SMALL MAMMALS IN URBAN LANDSCAPES: THE NEED FOR A MULTI-SCALED, MULTI-SPECIES APPROACH TO PLANNING AND MANAGEMENT

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6.1. ABSTRACT

As urban areas continue to expand and replace native ecosystems, it is becoming increasingly important that fauna assemblages and communities are conserved within the urban context. Terrestrial reptiles and small mammals, with limited dispersal abilities and specific habitat requirements, are particularly at risk of undergoing localised extinctions, yet these species are often overlooked by researchers, planners and managers in favour of the more obvious, iconic, and/or rare or threatened species. Based on the major findings from a case study conducted in

Brisbane, south-east Queensland, Australia, this paper presents six conservation guidelines for State, regional, and local urban planning frameworks and management strategies. The focus of the case study was native terrestrial reptiles and small mammals. The major findings showed that habitat attributes at multiple spatial levels were important for influencing reptile and small mammal assemblages. Accordingly, the guidelines promote a focus on: (1) Maintaining and increasing the amount of suitable habitat in the landscape; (2) Maintaining and increasing internal habitat condition (quality); (3) Maintaining and increasing connectivity between suitable habitat fragments; (4) Mitigating the placement and effects of roads in close proximity to habitat fragments; (5) Making habitat fragments as large and circular as possible; and, (6) Considering the whole landscape context and configuration, not just individual habitat fragments. These guidelines are directly relevant to landscape planning and management for reptiles and small mammals in lowland remnant habitat fragments of Brisbane's urban and urbanising landscapes, but are also relevant for local land managers in other Australian cities and internationally.

Key Words: Urban wildlife, guidelines, species richness, site, patch, landscape, collaboration.

6.2. INTRODUCTION

A common problem for urban planners and land managers is balancing the needs of humans and wildlife. That is, how can we achieve development with minimal environmental impact? Almost half the world's population currently resides in urban areas, and by 2030 this proportion is expected to reach 60% (United Nations, 2006). Humans, therefore, will be a predominantly urban species. The demand for space to accommodate new urban habitats for humans means the destruction, fragmentation, and degradation of natural habitats for many native fauna populations. Unfortunately, the characteristics that make a location suitable for urban development often coincide with areas that support high levels of biodiversity (Botkin and Beveridge 1997; Lugo 2002; Miller and Hobbs 2002), thereby producing very real and increasing threats to biodiversity conservation worldwide. Consequently, as natural ecosystems continue to be replaced by expanding urban settlements, it is increasingly important that native wildlife populations are conserved within the urban context.

Creating sustainable urban development is a complex challenge for researchers, urban managers, planners, developers, land holders, and the wider community (Johnson 1995; Adams et al. 2006; Fraser et al. 2006). Our ability to balance conservation and development is influenced by a multitude of social, political, economic, and ecological values, which often conflict (Johnson 1995; Campbell 1996; Ahern 1999). Decisions are often driven more by societal and economic demands, with science and ecological principles having less impact (Termorshuizen in press). We recognise that all these influences are important, yet if we are serious about successfully achieving truly sustainable urban environments, which effectively conserve native biodiversity, sound ecological science must be a primary influence underpinning decision-making.

Ecological knowledge is very important if we wish to efficiently allocate resources in an urban setting. To be of practical value, researchers should focus on investigating priority issues faced by planners and managers (Risser 1996). For example, knowing whether local-level attributes such as habitat quality, or patch-level attributes such as the size of a habitat patch, are more important will influence a trade-off between landscape restoration and habitat management. Similarly, spatially-explicit planning will be influenced by knowing whether patch-level attributes, such as patch size and shape, are more important than the structure of the whole landscape. Ecological researchers should recognise their important role in informing urban landscape managers and planners of the ecological principles which underpin biodiversity sensitive urban design, and effective and efficient management actions for conserving a range of habitat types and associated fauna assemblages (Hobbs 1997; Opdam et al. 2002). This will require researchers to investigate relationships between various species and habitat attributes operating across multiple spatial scales, and to compare these relationships, patterns, and process over time, and within various urban contexts. Urban landscape ecology can play a vital role in understanding the temporal changes and spatially-complex interactions between wildlife population dynamics, environmental patterns and processes, and urban landscape design and change. The challenge is to translate this information into decision support tools to underpin sustainable urban planning, design, and management (Moss 2000; Opdam et al. 2002; Oreskes 2004).

Actively addressing this challenge in urban landscapes for multiple species across multiple spatial scales is vital for rapidly urbanising areas, such as Australia's fastest urbanising region, south-east Queensland (Queensland Government 2005). Compared to other rapidly urbanising areas, such as the Netherlands, where "…spatial planning is widely accepted…and landscape values are becoming strong components in the competition for space…" (Opdam et al. 2002, pp. 777), Australia's planning frameworks lack a comprehensive and integrated spatial planning foundation (Peterson et al. in press).

The aim of this paper is to present a set of planning and management guidelines as one means for integrating ecological knowledge into planning frameworks and management strategies. These guidelines were based on the outcomes from a case study (Garden et al. in review b; in press) conducted for terrestrial reptiles and small mammals living in remnant forest fragments of Brisbane, Queensland. We begin by presenting an overview of Queensland's planning framework in order to set the context for the ensuing guidelines and enable comparisons between other urban landscapes, nationally and internationally. Next, we synthesise the design, analysis, and major outcomes of the Brisbane case study, and finally, we present the guidelines and a decision-support tree, which use the case study outcomes to prioritise urban planning and management objectives. We also outline recommendations to achieve these objectives.

6.3. QUEENSLAND PLANNING FRAMEWORK

Australia's political hierarchy is composed of Commonwealth (broadest sphere), State, and local governments. The Commonwealth is responsible for biodiversity conservation, particularly endangered species, through the *Environment Protection and Biodiversity Conservation Act 1999* (Commonwealth of Australia 2005). The Queensland *Integrated Planning Act 1997* (IPA) (Queensland Government 2006), is the foundation of the State's planning and development assessment legislation, which identifies a number of planning instruments to achieve its purpose "…to seek to achieve ecological sustainability…" (IPA s1.2.1). One important instrument is the requirement for each local government to develop and implement a planning scheme which will:

- identify ‘core matters’ (e.g., ecologically valuable areas);
- identify desired environmental outcomes for the planning scheme area; and,
- include measures that facilitate the achievement of the stated desired environmental outcome.

It is imperative that these planning instruments are informed by sound ecological knowledge derived from context-specific urban ecology research. In Queensland, while the *Vegetation Management Act 1999* (*VMA*, Queensland Government 2006) currently regulates the clearing of native vegetation, recent amendments to this legislation have revised the definition of an ‘urban area’ to exclude all rural-residential land. Further, the clearing of vegetation in urban areas remains largely exempt from assessment unless the vegetation is ‘endangered’ (i.e. < 10% of its pre-clearing extent remains), in which case it is assessed against the *VMA*. Consequently, remnant vegetation within the defined urban footprint of Brisbane (Queensland Government 2005) is given little legislative protection at the State level, with reliance being placed on local government planning schemes, which thus play a significant role in either protecting, or not protecting, important ecosystems and associated wildlife within the urban footprint.

Brisbane City Plan 2000 (Brisbane City Council 2002) has been approved under *IPA* as the planning scheme for guiding development in Brisbane. *Brisbane City Plan 2000* identifies the need for ecologically sensitive urban design and management within the urban footprint. Planning tools incumbent within the local planning scheme to guide planning and development assessments include:

- a strategic plan;

- development codes;
- development assessment tables; and,
- performance criteria.

However, within the urban footprint, many ecologically valuable habitats identified by State legislation, such as the *Regional Nature Conservation Strategy for South East Queensland 2003-2008* (Environmental Protection Agency 2006), are often ineffectively identified and protected by local planning schemes and associated tools due to a lack of knowledge and/or limited resources to enable detailed ecological assessments to be conducted at the local-level. Consequently, clearing and fragmentation of ecologically valuable areas and associated fauna assemblages continues within the urban footprint. There is, therefore, a need to improve integration between State, regional, and local strategies in order for urban development in the most rapidly urbanising regions to be sustainably and comprehensively planned and managed. To do so, it is imperative that: (1) State, regional, and local-level planning schemes accurately identify and actively protect habitats of State, regional, and local significance; and (2) local-level planning and management decisions consider broader landscape patterns and processes. To facilitate this decision-making, spatially explicit, multi-scaled and multi-species knowledge, derived from sound urban ecology research, is required and must further be reliably and clearly communicated to decision-makers (Dale et al. 2000; Villard 2002).

6.4. BRISBANE CASE STUDY

The case study was conducted within Queensland's capital city of Brisbane (153°02'E, 27°28S; area 1,220 km^2; population ~ 1 million). Brisbane is Australia's third largest capital city, and is located in the coastal south-east of the State (Figure 6.1), which is Australia's most rapidly developing region with an average annual population increase of approximately 55 000 people (Queensland Government 2005). More than one third of this population resides within the local government area of Brisbane (Queensland Government 2005), making the city Australia's most rapidly urbanising capital city (Brisbane City Council 2002). The popularity of Brisbane is due in large part to its sub-tropical climate and high quality of life. The city area supports a rich native biodiversity due to its location within the McPherson-Macleay Overlap zone (Burbidge 1960; Queensland Government 2003). This zone is characterised by the merging of the longitudinal extremities of two of Australia's major phytogeographic regions – the northern tropical (supporting Torresian species) and the southern temperate (supporting Bassian species) – and so is characterised by exceptionally high floristic and faunal diversity representative of these two regions (Burbidge 1960).

Clearing of remnant vegetation within the local government area has been extensive, with little more than a third of the pre-settlement original woody vegetation cover remaining (Catterall and Kingston 1993; Environmental Protection Agency 2006). The term 'remnant' refers to woody vegetation where the "…dominant canopy has greater than 70% of the height and greater than 50% of the cover relative to the undisturbed height and cover of that stratum and dominated by species characteristics of the vegetation's undisturbed canopy…" (*VMA*, Queensland Government 2006). The majority of Brisbane's remnant vegetation exists as contiguous forest in the D'Aguilar Ranges to the city's west due to unsuitable topography which prevented urban or agricultural development in the past. In comparison, more than 80%

of the original lowland forests (< 100 m ASL) have been cleared for urban and agricultural development, resulting in highly fragmented lowland remnant vegetation patches that vary in their internal condition, size, shape, isolation, context, and disturbance and management histories (Catterall and Kingston 1993; Environmental Protection Agency 2006).

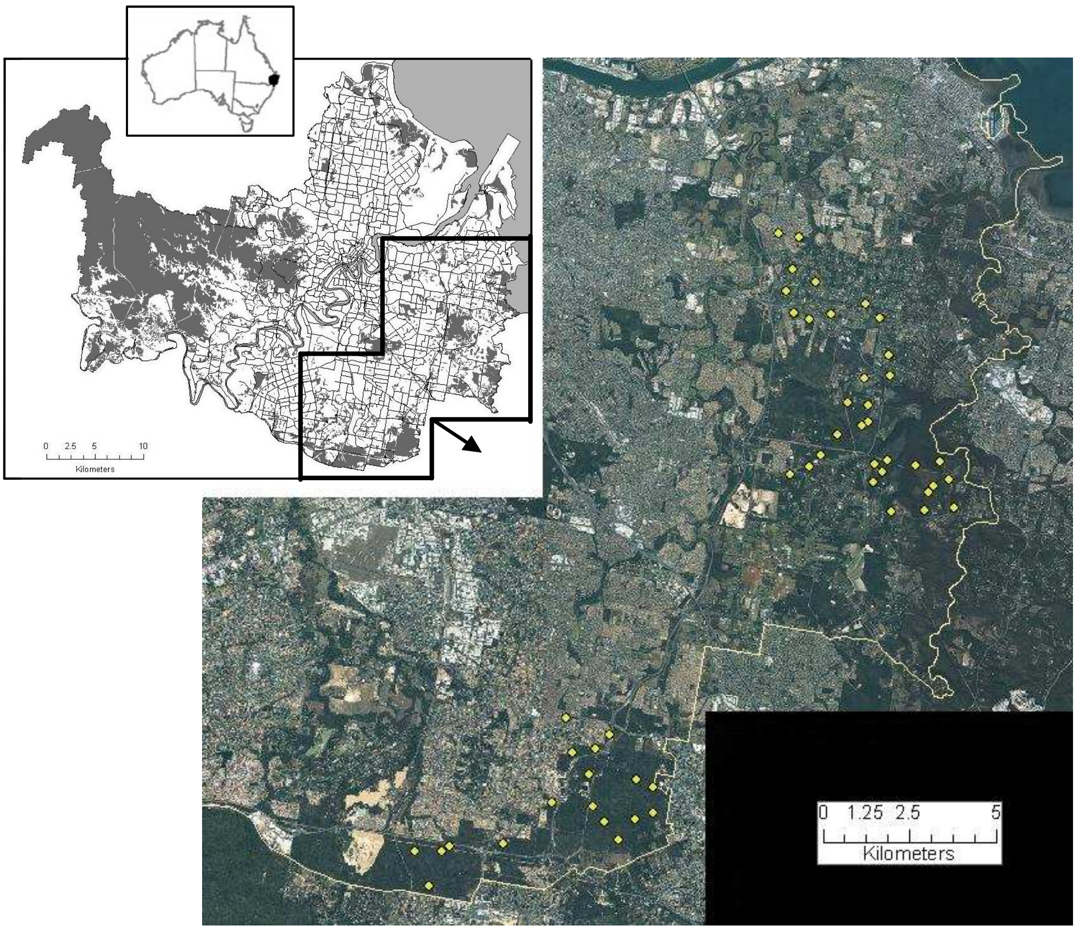

Figure 6.1. Location of the Brisbane case study area in the south and south-east peri-urban suburbs of the Brisbane City local government area. Quickbird satellite imagery from 2006 shows the location of fauna survey sites (○) within remnant fragments in the peri-urban zone. The pale line indicates the Brisbane city council local government area boundary, with Redland Shire Council and the coastline to the east and Logan Shire Council to the south.

The majority of this remaining lowland vegetation, including the largest of the remaining lowland remnant fragments, occurs in the peri-urban fringe of Brisbane's south and south-east suburbs. 'Peri-urban' is used here to describe areas subject to rapid urbanisation, often occurring at the juxtaposition of high density urban development and rural land-uses. These remnant fragments are of high conservation value for the ecosystems they represent and associated wildlife populations, many of which are not found in the western ranges or lowland wetland areas. Accordingly, many of these lowland remnant fragments have been identified as areas of nature conservation significance in the *Regional Nature Conservation Strategy for South East Queensland 2003-2008* (Queensland Government 2003).

A major challenge then, within Brisbane, is how to plan and manage urban and peri-urban landscapes so that native biodiversity is conserved without negatively influencing opportunities for urban growth. Brisbane has the advantage over other cities in that it has a single local government council body, Brisbane City Council (BCC), governing the whole city. This allows planning decisions and management actions to be applied across the greater metropolitan area. It is therefore important that BCC has a sound scientific basis to help develop and improve its adoption of ecologically sustainable planning, management, development, and restoration decisions, thereby maximising the potential for long-term protection of the city's and the region's native biodiversity.

6.4.1. Synthesis of Study Findings

Our Brisbane case study focussed on terrestrial reptiles and small mammals inhabiting lowland remnant vegetation fragments within the BCC local government area (Figure 6.1).

Many of these species do not occur within the highly-modified inner suburbs (< 10% native vegetation remaining) as they are highly sensitive to habitat disturbances because of their specific habitat and/or dietary requirements, and limited dispersal abilities. They are largely forest-dependent and so occur primarily where forest cover is available for shelter and ensuring low-risk dispersals. In urban-dominated landscapes, terrestrial locomotion and dependence on cover often limits the dispersal success of these ground-dwelling species, particularly in comparison to birds, for instance, which are capable of flying between habitat fragments. Terrestrial reptiles and small mammals are therefore at a high risk of undergoing localised extinctions if their habitat requirements and sensitivities are not incorporated into urban planning and management decision-making processes.

Fauna surveys of 51 sites located within 32 remnant forest fragments were conducted in the spring/summer seasons of 2004 and 2005 (Garden et al. in review a). Habitat assessments of each site recorded the structural complexity, vegetation composition, and disturbances at the local-level. Patch-level attributes, such as the shape and size of forest fragments, and landscape composition and spatial configuration were obtained using GIS data and high resolution Quickbird satellite imagery.

The fauna surveys detected a total of 19 reptile species and nine small mammal species (Table 6.1). Habitat attributes at each spatial level were found to be important for influencing species assemblages, although the relative importance of attributes within each spatial level varied between reptile and mammal assemblages (Table 6.2). Overall, reptiles were most influenced by landscape composition and configuration as well as local-level habitat structure and composition; whereas, mammals were most influenced by landscape composition, followed by local-level habitat structure and composition, and then patch size and shape (Table 6.2) (Garden et al. in review b, in press). These findings highlight the need for researchers to investigate the responses of multiple species and fauna assemblages to attributes operating across multiple spatial levels. Studies that investigate only a single spatial level (e.g., local-level) or single-species/taxa may mask or exaggerate the influence of patterns and processes occurring at other levels (Wiens 1994; Hobbs 1999) and their influences on other species.

Table 6.1. Cumulative list of native reptiles and small mammals detected during the Brisbane case study fauna surveys.

	FAMILY GROUP	SCIENTIFIC NAME	COMMON NAME
Native Reptiles	Agamidae	*Diporiphora australis*	Tommy round-head
		Physignathus lesuerii	Eastern water dragon
		Pogona barbata	Bearded dragon
	Colubridae	*Dendrelaphis punctulata*	Common tree snake
	Elapidae	*Pseudechis porphyriacus*	Red-bellied black snake
	Gekkonidae	*Diplodactylus vittatus*	Eastern stone gecko
	Pygopodidae	*Lialis burtonis*	Burton's snake-lizard
	Pythonidae	*Morelia spilota*	Carpet python
	Scincidae	*Anamalopus verreauxii*	Verreaux's skink
		Calyptotis scutirostrum	Scute-snouted calyptotis skink
		Carlia foliorum	Tree-base litter-skink
		Carlia pectoralis	Open-litter rainbow skink
		Carlia vivax	Storr's rainbow skink
		Cryptoblepharus virgatus	Fence skink
		Ctenotus taeniolatus	Copper-tailed skink
		Eulamprus quoyii	Eastern water skink
		Lampropholis amicula	Secretive skink
		Lampropholis delicata	Garden skink
	Varanidae	*Varanus varius*	Lace monitor
Native Mammals	Dasyuridae	*Antechinus flavipes*	Yellow-footed antechinus
		Antechinus subtropicus	Subtropical antechinus
		Planigale maculata	Common planigale
		Sminthopsis murina	Common dunnart
	Muridae	*Melomys* sp.	Unknown species
		Rattus fuscipes	Bush rat
		Rattus lutreolus	Swamp rat
	Peramelidae	*Isoodon macrourus*	Northern brown bandicoot
		Perameles nasuta	Long-nosed bandicoot

Table 6.2. Summary of the key habitat attributes, at each spatial level, for reptiles and small mammals in Brisbane's lowland remnant habitat fragments. These attributes were characteristic of survey sites that were found to support a high diversity of reptile and/or small mammal species.

SPATIAL LEVEL	REPTILES	MAMMALS
Landscape **(100s – 1000s ha)**	• High degree of forest patch connectivity within 5 km of survey sites • Predominantly forest habitat within 1 km of survey sites • Larger forest mean patch area within 4 km of survey sites • Low proportion of road reserves within 500 m of survey sites	• Predominantly forest habitat within a low density/rural habitat context within 5 km of survey sites
Patch **(1 – 100s ha)**	• N/A	• Larger forest patch size • Less-linear forest patch shape
Local **(< 1 ha)**	• No-low weed cover • Low soil compaction • Large amount of fallen woody material • Presence of termite mounds	• Large number of grass trees • Large number of paperbarks • Low to moderate soil compaction

6.5. MANAGEMENT GUIDELINES

In this section we briefly discuss the role of guidelines for urban planning and management. Then we list six guidelines and explain how the objectives for each guideline may be achieved, by providing practical examples and recommendations for urban planning and management. For each guideline, we also provide a brief ecological rationale based on the findings from our Brisbane case study, and other national and international ecological studies.

Given the dynamic and novel influences of urban-based disturbances on natural environments, coupled with species-specific disturbance sensitivities and habitat requirements, the effects of urbanisation are difficult to translate into simple management rules (Villard 2002). It is therefore important that planning and management decisions are based on rigorous ecological knowledge in order to promote decisions that will mitigate detrimental long-term or broad-scale ecological impacts (Dale et al. 2000). A first step in making informed and ecologically-sensitive decisions is a comprehensive understanding of species occurrences within remnant fragments, their distributions across the landscape, and the specific habitat requirements of those species most sensitive to habitat disturbances. This ecological knowledge will enable decisions to focus on avoiding or minimising development in priority conservation areas, and designing and managing urban areas so that important habitat elements necessary for the survival of sensitive species are protected and sustainably managed. Such disturbance-sensitive species may currently persist in urban fragments with little management attention, and so are often overlooked by researchers, planners, and managers, in favour of threatened species. However, management decisions made only with consideration for threatened species are unlikely to be effective for conserving the suite of urban-sensitive species, particularly when these urban-sensitive species have vastly different life history traits (e.g., koala, *Phascolarctos cinereus*, versus common dunnart, *Sminthopsis murina*). Further, many urban-sensitive species populations are also experiencing gradual declines in abundance. Without proactive consideration of the habitat requirements of urban-sensitive species, many are likely to undergo localised extinctions in the future, making today's urban-sensitive species tomorrow's threatened species.

It is therefore vital that the local status (i.e. stable, increasing, or declining) of sensitive-species populations occurring within an urban area is accurately determined, and decisions are prioritised to prevent declines in the long-term viability of these populations. To do so comprehensively requires long-term monitoring programs operating over 10-20 years, as many species exhibit time lags in their responses to habitat disturbances (Tilman 1994; Hanski 1998; Carlson 2000; Possingham and Field 2001). However, short-term research is equally important, providing information for determining priority habitat attributes for species, and formulating guidelines for future planning, and current and future management of urban landscapes.

The following guidelines provide an example of how ecological knowledge may be integrated into urban planning and management decision-making processes. These guidelines are based on the findings from our Brisbane case study and so identify priority multi-scaled habitat attributes for terrestrial reptiles and small mammals that must be considered in the decision-making processes (Garden et al. in review b, in press). Overall, the guidelines are applicable to State, regional and local levels of planning and management, although they are particularly important for underpinning planning frameworks that regulate future urban development, as these frameworks identify areas of ecological significance, and manage the location, scale, and form of urban development. Specific guidelines for addressing key attributes at different spatial levels, however, may best be addressed by planning and/or management at different spatial levels (Figure 6.2).

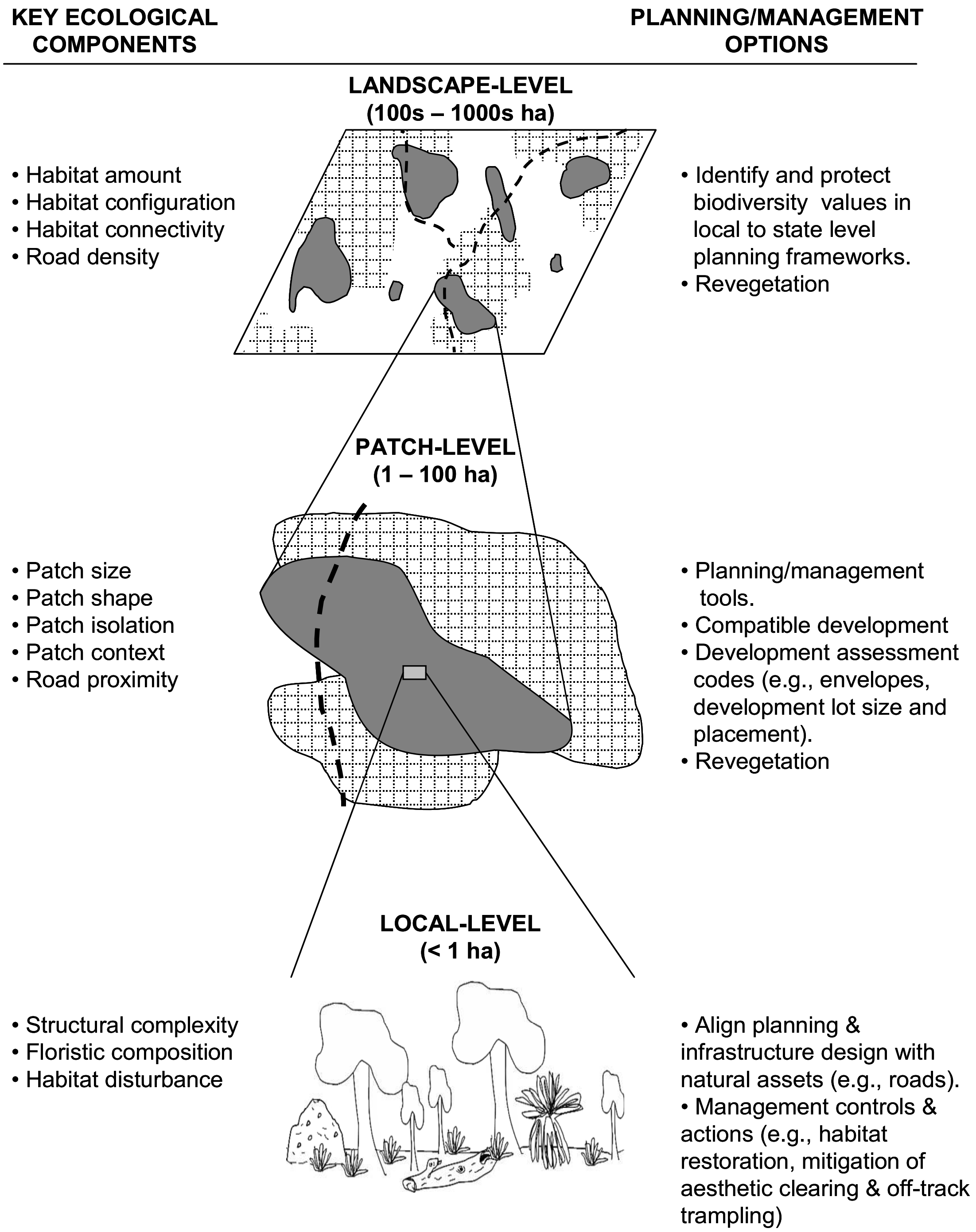

Figure 6.2. Integrating ecology, planning, and management at multiple spatial levels. Left-hand column = multi-level key ecological components for terrestrial reptiles and small mammals, determined from the Brisbane case study; Right-hand column = examples of planning and management actions for addressing the key components at each spatial level. The landscape-level and patch-level schematics show remnant fragment/s located within various land-uses (■ remnant forest, ⊞ urban, □ rural), and across jurisdictional boundaries (dotted lines). The site level schematic shows habitat with high vertical and horizontal structural complexity.

Each of the ensuing six guidelines are presented in four parts: (i) the principal target instrument (e.g., State, regional, or local planning or management) is identified for each guideline; (ii) the objective for addressing the guideline is identified; (iii) examples of planning and/or management actions that could achieve the guideline objective are provided; and, (iv) the ecological rationale underpinning the guideline. To facilitate prioritisation of conservation actions and funding allocation, the guidelines are presented in order of relative importance (highest priority first) for reptiles and small mammals, as concluded from our Brisbane case study. In addition, a decision-support tree linked to the guidelines, presents a hierarchical series of questions which may be easily incorporated into planning and management decisions (Figure 6.3). The guidelines and decision-support tree are directly applicable for conservation planning and management in Brisbane, but are also suggested as a suitable framework for facilitating the integration of other ecological research findings on various species within different urban landscapes, into planning and management strategies.

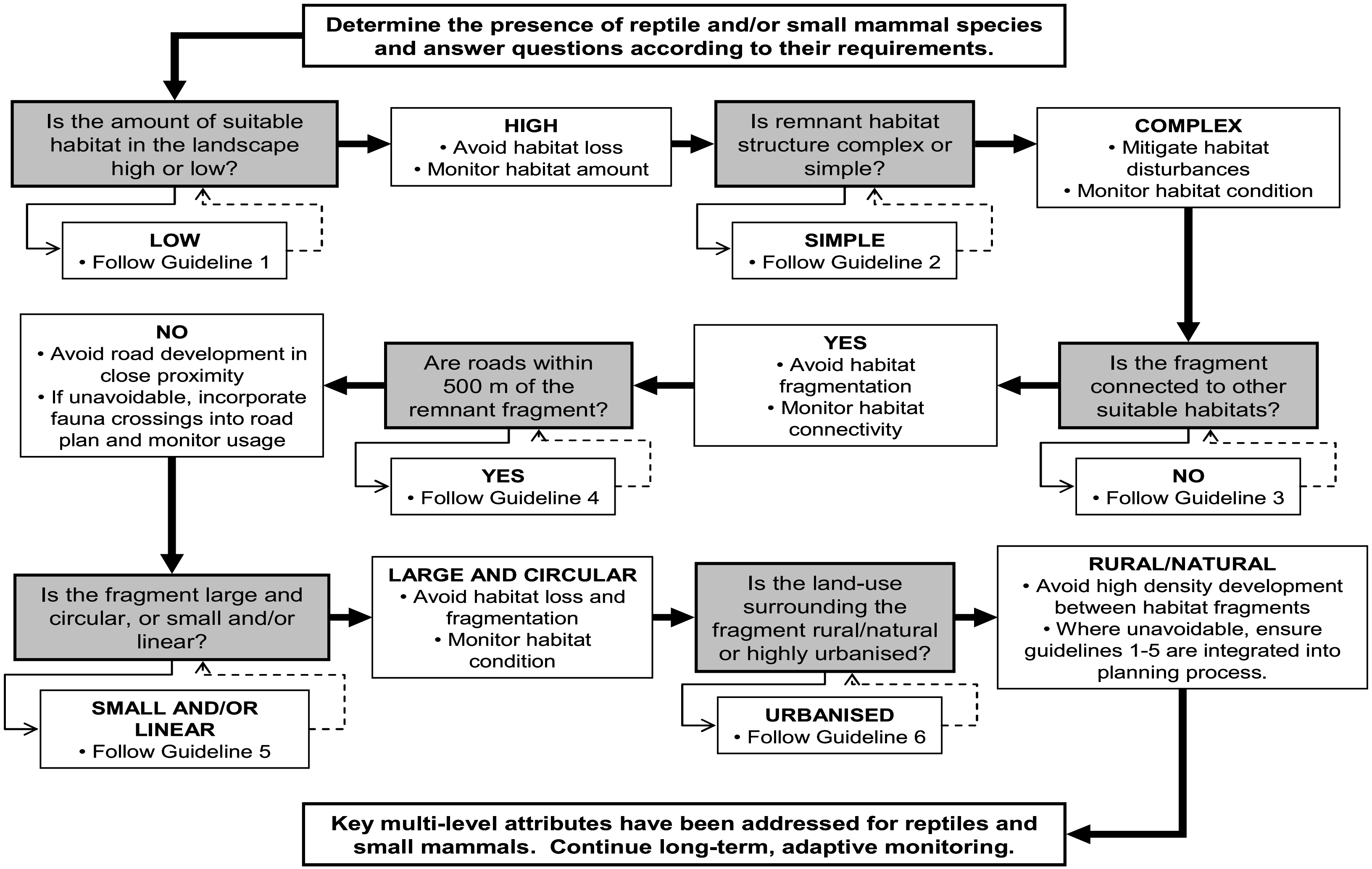

Figure 6.3. Decision-support tree for planning and management. Dotted lines indicate re-evaluation actions. In an ideal, ecologically sensitive urban landscape, answers to each question would directly follow the path of the thickest arrows.

Guideline 1: The amount of suitable habitat in the landscape is the most important consideration.

Target Instrument: State, regional, and local planning and management strategies.

Objective: To maintain and/or increase the amount of suitable habitat within the landscape. Suitable habitat, in this sense, refers to high quality habitat that fulfils the habitat and resource requirements of sensitive reptiles and small mammals (see Guideline 2).

Actions: Suitable habitats of high ecological value should be identified and protected in State, regional, and local planning frameworks. The identification of such habitats should occur in collaboration with ecological researchers to ensure multiple habitat requirements for different species are addressed. These valuable habitats need to be allocated appropriate land-use classifications (e.g., conservation or protection areas), which only allow development that is compatible with the conservation of these values. This will require planning tools to identify appropriate outcomes at multiple scales, and design specifications that are appropriate at the local scale. At the level of the local government, a variety of planning tools may be utilised to maintain and increase suitable habitat including: development codes, tradeable development rights, covenants, mitigation banking, and voluntary conservation agreements. The selection of appropriate tools will depend on several factors, such as the level of certainty that is required, the cost, the social and political acceptability, equity, and ease of enforcement (Gunningham and Young 1997). It is important that ecological knowledge is also a key influence in selecting appropriate planning tools.

Local landscape management strategies are also important for fulfilling this guideline in already developed landscapes. Appropriate revegetation and rehabilitation activities on public and private lands will increase the amount of suitable habitat in the landscape. Ecological knowledge will be important for identifying priority activities and target areas for management strategies that will produce a more effective and resource efficient conservation outcome. Community support and cooperation for such management strategies may best be facilitated through education campaigns that highlight the benefits to human health and quality of life by adopting ecologically sustainable urban management. Appropriate incentives should also be used in conjunction with education campaigns to gain support from community members and encourage appropriate activities on privately owned land. If carefully planned, revegetation and rehabilitation activities may address multiple guideline requirements, such as increasing patch size and reducing edge effects (Guideline 5), increasing connectivity (Guideline 3), and enhancing habitat condition (Guideline 2), which will help to minimise additional costs associated with tackling these issues independently.

<u>Rationale:</u> The Brisbane case study showed that native mammal and reptile species richness was higher in forest fragments for which a high amount of suitable similar habitat existed within the broader landscape. Clearing habitats decreases the availability of critical resources for various species, which inhibits the short-term survival of individuals and the long-term viability of populations. The impacts of habitat loss are well documented in the scientific literature for a wide variety of species, such as koalas and gliders in Australia (McAlpine et al. 2006a; McAlpine and Eyre 2002), butterflies in the United Kingdom (Wood and Pullin 2002), white-backed woodpeckers in Europe (Carlson 2000), bird communities in Malaysia (Soh 2006), and reptile, mammal, and bird assemblages in Madagascar (Scott et al. 2006). The

sensitivities of species to habitat loss, however, are highly variable among species and across different landscapes (Andrén 1994; Bascompte and Solé 1996; Pearson et al. 1996; With and King 2001; Fahrig 2003; Scott et al. 2006). It is therefore essential that sound ecological knowledge underpins management and planning decisions, and that this knowledge is context-specific.

Guideline 2: The condition (quality) of habitat is the second most important consideration.
Target Instrument: Local-level planning and management strategies.

Objective: To maintain the internal condition of undisturbed habitat fragments and increase the internal condition of degraded habitat fragments.

Actions: The internal condition of habitat fragments must be assessed prior to making planning and management decisions that will influence those habitat fragments. Local government planning schemes must aim to identify and protect important elements of habitat structure and composition. Prohibiting high-density development and ensuring compatible low-density development is of high priority where habitat fragments have high structural complexity (Figure 6.4). In these situations, planning designs (e.g., road/building design and location) must be required to maintain natural assets of ecological importance (e.g., termite mounds, fallen woody material, and grass trees). This will require planning designs such as: low density, clustered developments in more valuable areas; high density clustered development in areas of low environmental significance; and, the regulation of building location envelopes (which dictate the location and limitations of structures) that protect structurally complex vegetation at the lot scale by limiting structures to appropriate sites on each allotment. Priority management actions to maintain high structural complexity in habitat fragments include mitigating the effects of human habitat misuse (e.g., off-track trampling, aesthetic clearing, and garden waste dumping) that degrade structural elements, and investigating appropriate fire regimes that protect human life and property, yet are not so frequent or intense that key structural elements are degraded.

Fragments with low structural complexity (Figure 6.4), such as those degraded by human land-use and past developments should be rehabilitated to increase the complexity of important horizontal (e.g., fallen woody material and coarse leafy ground cover) and vertical structures (e.g., understorey, midstorey, and canopy layers). This may best be achieved through revegetation guidelines and ongoing management actions that mitigate human habitat misuse, as described previously. In addition, planning codes and guidelines for future public parks and gardens, and private land holdings should stipulate the use of native plantings and natural components that will increase structural complexity. Existing public parks and gardens should also be assessed for their structural complexity and retrofitted appropriately where complexity is low. Private land holders must also be informed of the importance of structural complexity and encouraged, by using appropriate incentives and education programs, to manage their own gardens and backyards accordingly. Appropriate planning and management of modified green space areas is particularly important when these areas are adjacent to remnant habitat fragments.

Figure 6.4. Structural habitat complexity at selected survey sites from the Brisbane case study. Left-hand column = examples of sites with high structural complexity (native reptiles and small mammals detected); Right-hand column = examples of sites with poor structural complexity (no native mammals and only two native reptiles detected).

Rationale: The findings from the Brisbane case study indicated that local-level habitat structure and composition were very important for reptiles and mammals, with structural elements being more important than compositional elements. Fauna survey sites that were structurally complex supported a higher diversity of native terrestrial reptile and small mammal species. These findings are consistent with previous ecological research that have posited the importance of maintaining high habitat quality for the persistence of bird, mammal, and reptile assemblages (Fischer et al. 2003; Jellinek et al. 2004; Kanowski et al. 2006), with the relative importance of various structural and compositional attributes varying according to the species of interest (Wood and Pullin 2002). The maintenance and successful restoration of high quality habitats will also depend on identifying and mitigating anthropogenic disturbances that may degrade habitat quality (Lindenmayer et al. 1998).

Guideline 3: The third most important consideration is to maintain and increase connectivity between forest fragments.

Target Instrument: State, regional, and local planning management strategies.

Objective: To maintain and increase connectivity between habitat fragments that will facilitate movements by dispersal-limited and disturbance-sensitive species.

Actions: Ideally, continued clearing and fragmentation of valuable remnant vegetation must be stopped. As this is not always possible, appropriate connectivity strategies must be incorporated into future planning strategies. Connectivity strategies may include: continuous vegetated wildlife corridors (Bowne et al. 1999; Pirnat 2000); vegetated 'stepping stones' within a suitable matrix (Haddad 2000; van Langevelde et al. 2002; Baum et al. 2004); and, artificial structures such as road over- and under-passes (Taylor and Goldingay 2003; Clevenger 2005). As the appropriate connectivity strategy, design, and subsequent functionality in enabling low-risk species dispersals will vary for different species and within different land-use contexts, it is important that connectivity decisions are based on local fauna and habitat assessments coupled with ecological knowledge of species dispersal abilities, habitat requirements, and sensitivities (Dickman and Doncaster 1989; Bowne et al. 1999; Lidicker and Peterson 1999).

In already fragmented urban landscapes, options to restore connectivity must be investigated by local managers. This is likely to require the cooperation of private land holders, located in the matrix between priority habitat patches, to manage backyards and gardens to achieve a combined, low-risk connection suitable for certain species (Rudd et al. 2002). Successful connectivity through private properties will require substantial government practical support, such as community education workshops and native plant giveaways, as well as ongoing guidance and advice, and targeted incentives. In addition, artificial connections across and/or under roads that bisect priority habitat patches must be investigated and implemented where possible.

<u>Rationale:</u> In urban and peri-urban landscapes, habitat fragments are often isolated by the built matrix and associated infrastructure. This prevents natural dispersals by many reptile and small mammal species, increasing the demand on limiting resources within fragments, and limiting genetic flow between fragments, which ultimately influences the long-term viability of wildlife populations across the landscape. The Brisbane case study showed that the

occurrence of native reptiles in remnant fragments was positively influenced by a high degree of connectivity between fragments in the broader landscape. This supports previous studies that have also shown the need to maintain and increase landscape connectivity for various species (Soule 1991; Ferreras 2001; Lindenmayer et al. 2006; With et al. 2006).

Achieving connectivity across various land-uses will require partnerships between decision-makers, researchers, and private land holders. An example of such successful collaboration between decision-makers and researchers is the Compton Road upgrade project in Brisbane. An integral component in the planning and design of this road upgrade was the incorporation of a combination of artificial crossings and associated infrastructure, such as fencing (Figure 6.5), designed to promote low-risk species dispersals between the ecologically significant Karawatha Forest on one side of the road, and Kuraby bushland on the other side (Bond and Jones 2006). These fauna crossings and associated structures successfully reduced previously high incidents of road-kill and also facilitated low-risk dispersals by a variety of terrestrial and arboreal species (Bond and Jones 2006). This example highlights the ability for planners, managers, and ecologists to work together to achieve an outcome that supports urban progress and is also ecologically sensitive.

(a)

(b)

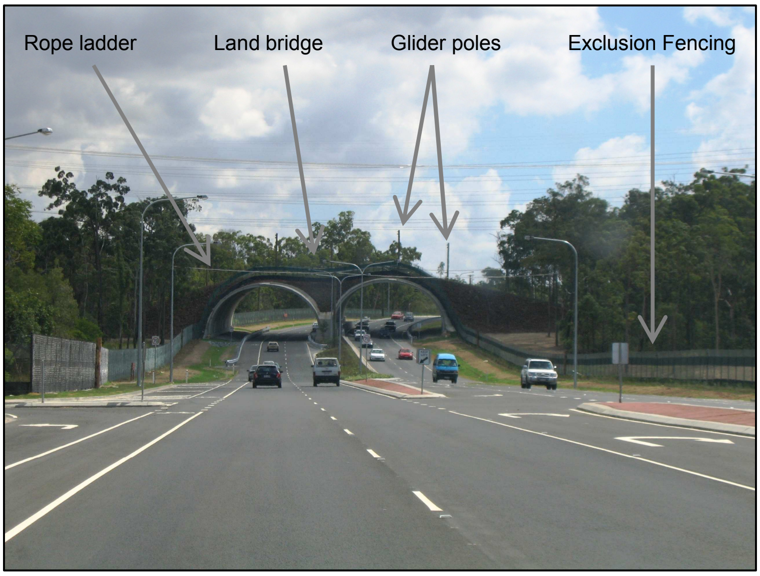

Figure 6.5. Compton Road, Brisbane: (a) the original two-lane road at the start of the upgrade, and (b) the final four-lane road following the upgrade. Both photos, taken at approximately the same position on Compton road, show Karawatha forest reserve on the left and Kuraby forest on the right. The post-upgrade photo in (b) shows one of the rope ladders located at various points across the road for crossings by arboreal species such as possums, the specialised road-side exclusion fencing to prevent fauna road mortalities, and the landscaped fauna land-bridge with fencing and glider poles along its length. Not visible in the photo are two designated fauna underpasses to facilitate crossing by small terrestrial fauna species.

Guideline 4: Avoiding the impact of roads on forest fragments is the fourth most important consideration.

Target Instrument: State, regional, and local planning and management strategies.

Objective: To minimise the negative influence of roads on habitat fragments and associated native species assemblages.

Actions: Urban planning decisions at all levels of government must avoid placement of roads, or increased traffic volume on existing roads, adjacent to or within close proximity of high quality remnant fragments. Based on ecological knowledge, planning schemes and incorporated codes/policies must also identify design options, such as road widths, curvatures, fauna crossings, and signage, that will protect the value of remnant vegetation, minimise the impact of roads on wildlife, and retain the natural characteristics of an area. Where roads already exist in close proximity to remnant fragments, or where their construction or upgrade is inevitable, managers and planners must work with researchers to incorporate appropriate fauna crossings (see Guideline 3 and Figure 6.5) into the road design so that the impacts on species are minimised.

Rationale: The Brisbane case study found a negative relationship between native reptile species richness in forest fragments and the density of roads in close proximity (within 500 m) to fragments. Roads bisect the landscape, fragment habitat patches, and present significant dispersal barriers to species living within habitat fragments (Andrews 1990; Forman 1999; Ramp et al. 2006). The negative direct effects of roads on fauna assemblages as well as associated indirect impacts, such as pollution and altered habitat quality, are broadly recognised in the ecological literature (Quarles et al. 1974; Reijnen and Foppen 1994; Forman 1999; Dique et al. 2003; Hawbaker et al. 2006). Despite this, little is done to mitigate the impact of roads in urban areas. As a major component of urban environments, the impact of roads on remnant habitats and species assemblages must be carefully considered, managed, and mitigated, wherever possible.

Guideline 5: The fifth most important consideration is to make forest fragments as large and non-linear as possible.

Target Instrument: State, regional, and local planning and management strategies.

Objective: To manage forest fragments so that they are as large and non-linear as possible.

Actions: Urban planning and management need to maintain and promote large, non-linear fragments, prevent the creation of small and/or linear fragments, and require future planning to incorporate vegetated buffers around existing remnant fragments. Comprehensively achieving this action will require integration across local, regional, and state levels, with local-level actions supporting state wide assessments of biodiversity values. When planning new developments, it is vital that planners avoid developments that will clear remnant native vegetation. As this is not always possible due to existing and future development rights, it is important that planners promote urban consolidation (e.g., high density housing) rather than acreage development (e.g., ¼ acre blocks) (see Guideline 1). However, where the public demand for acreage development is high, unconsolidated development should occur only in less environmentally-sensitive areas. In developed areas, remnant fragment size and shape may be manipulated by encouraging appropriate management of public and private gardens that are adjacent to remnant fragments. This will require following similar revegetation and garden management actions outlined in Guidelines 1 and 3. In peri-urban zones, and where possible in urban areas, revegetation projects that increase fragment size and decrease linearity will benefit wildlife species.

Rationale: The shape and size of forest fragments in the Brisbane case study had an impact on mammal richness (Garden et al., in review b), with richness being higher in larger and more circular (less linear) fragments. These findings support previous studies, such as Wiens (1994), which conclude that larger patches support more species. These findings are also indicative of edge effects. The magnitude of edge effects on species is related to the contrast between the habitat edge and the adjacent land-use (Lindenmayer and Fischer 2006). Compared to their non-urban counterparts, edge effects in urban settings are intensified as the boundary between landscape types is often more abrupt and involves novel, man-made structures, surfaces, and intensively managed vegetation (Adams et al. 2006). Edge effects are important as they alter species compositions by influencing the flow of organisms between fragments, and facilitating the incursion of disturbances and exotic pests, competitors, and predators, into core habitat areas (Rutledge and Lepczyk 2002). Large and robust fragments, for instance, will contain edge and core habitats, and so support a variety of species associated with these different habitat types. Small and/or linear fragments, however, are more likely to be composed of edge habitat only, and so will support lower species richness. Given species-specific responses to patch size and edge effects, it is important that management and planning decisions are underpinned by sound scientific research regarding the habitat requirements and sensitivities of multiple species.

Guideline 6: Considering the whole landscape and not just patches is the next most important consideration.

Target Instrument: State, regional, and local planning and management strategies.

Objective: To promote aggregated habitat fragments and manage the urban matrix for the benefit of wildlife species.

Actions: The spatial arrangement of fragments within the landscape, and the landscape surrounding remnant fragments, must be managed appropriately for fauna. This requires researchers, planners, and managers rejecting the binary landscape view, which considers landscapes to be composed of habitat patches within a non-habitat matrix. Instead, landscapes must be investigated, planned, and managed as a spatially complex mosaic of habitat types that vary in their permeability and functional use for different species (Danielson 1991; Lidicker and Peterson 1999; Fahrig 2002; Lindenmayer et al. 2003; Opdam et al. 2003). Achieving these actions will require integration across state, regional and local levels, and will also require decision-making processes regarding appropriate spatial configurations and contexts of fragments for different species being underpinned by ecological knowledge.

Rationale: The findings from the Brisbane case study showed that the configuration and context of habitat fragments was important for native species richness, implying that the ability to disperse is important here for mammals and many of the reptiles (Watson et al. 2005). This highlights the importance of planning and managing urban landscapes so as to maintain an appropriate spatial arrangement of habitat fragments. In addition, native mammal species richness was positively associated with the proportion of the landscape comprised by rural land-use, and both reptiles and mammals were negatively associated with the amount of built habitat in the landscape (Garden et al. in review b). These findings support previous research that also demonstrates that the matrix matters, and that the location of forest remnant fragments within this varied matrix is important for various species (Andrén 1994; Friesen et al. 1995; With and King 2001; Brotons et al. 2003; Riffell et al. 2003; Dunford and Freemark 2004; Maiorano et al. 2006). Consequently, planning and management must consider the matrix as well as the habitat fragments. In particular, it is important to avoid urban intensification between large, high quality patches, as this would decrease dispersal success by different species.

6.6. CONCLUSION

Today's landscapes are the legacy of yesterday's land-use planning and management decisions, just as tomorrow's landscapes will reflect our current uses, management successes and failures (Marucci 2000). Urban development and expansion will continue with or without the input of ecological research. However, if we are serious about successfully conserving biodiversity for the long-term, in the face of urban expansion, urban planners and managers require scientifically sound knowledge and "…tools to understand their cities and regions as…environmental systems that are part of regional and global networks…" (Campbell 1996, pp. 306).

The challenge of sustainable development is complicated by the numerous private stakeholders and various administrative units who make independent planning and management decisions often with little regard for influences on and from areas beyond the boundary of their focal area. As fauna species do not recognise anthropogenic boundaries, successful urban planning and management will also require cross-boundary collaboration and cooperation between local government authorities, and between local governments and private/public sectors (Stubbs et al. 2000; Woinarski and Fischer 2003; van der Ree and McCarthy 2005; McAlpine et al. 2007; Plumptre et al. 2007). Encouraging the cooperation of private sectors is likely to require targeted education programs, and appropriate government support and compensation (Christensen et al. 1996; Marzluff and Ewing 2001; McAlpine et al. 2007).

The long-term success of ecologically-sensitive development and management is dependent not only on a multi-species, multi-level, collaborative, and cooperative approach, but also on our ability to learn and adapt from our past successes and failures, and our willingness to try novel approaches to urban design and management. Such adaptive management is an ongoing process that is refined over time with accumulated knowledge and continual learning, and is essential for fruition of the ultimate goal in urban areas; that of increased development with decreased ecological impacts.

(a) Scute-snouted calyptotis skink (*Calyptotis scutirostrum*)

- Family: Scincidae
- Fossorial, shade-loving skink.
- Average size: SVL 55 mm.
- Distribution: South-east Queensland,north-east New South Wales.
- In Brisbane: common in gardens and bushland with suitable habitat.

This photo shows an adult calyptotis skink on my hand.

(b) Tree-base litter skink (*Carlia foliorum*)

- Family: Scincidae
- Secretive, sun-loving skink.
- Average size: SVL 39 mm.
- Distribution: Queensland, New South Wales.
- In Brisbane: restricted to suitable remnant bushland habitat in peri-urban landscapes.

This photo shows a breeding male on my hand, note the pink-orange colour of the tail.

(c) Open-litter rainbow skink (*Carlia pectoralis*)

- Family: Scincidae
- Active, sun-loving skink.
- Average size: SVL 52 mm.
- Distribution: Queensland.
- In Brisbane: restricted to suitable remnant bushland habitat in peri-urban landscapes.

This photo shows an open-litter rainbow skink on my hand – most likely a female.

(d) Fence skink (*Cryptoblepharus virgatus*)

- Family: Scincidae
- Swift, agile, sun-loving skink.
- Average size: SVL 40 mm
- Distribution: Queensland, New South Wales, South Australia, Western Australia.
- In Brisbane: Ubiquitous, thrives in urbanised landscape, favours brick/stone walls, pavements, and paling fences.

This photo shows a fence skink on my hand.

Plate 7. Skinks: (a) Scute-snouted calyptotis skink; (b) Tree-base litter skink; (c) Open-litter rainbow skink; (d) Fence skink.

Chapter 7

GENERAL DISCUSSION

Like many other sub-tropical regions of the world, Brisbane is an area naturally high in native fauna species diversity and endemism, and is also currently undergoing rapid landscape transformation due to urban development. If we are to successfully conserve our native fauna in the face of rapid urban expansions, conservation within the urban landscape must be a reality. This thesis provides new information about the ecology and management of native wildlife in fragmented urban landscapes. In this final chapter, I briefly synthesise the major findings discussed in detail in earlier chapters and highlight how these findings contribute to ecological knowledge. I then identify implications for the conservation of reptiles and small mammals in urban landscapes, based on the project's findings. Finally, I outline limitations to the project and use these to suggest recommendations for future research.

7.1. OVERVIEW

The primary aim of this project was to determine the relative importance of local, patch, and landscape-level habitat attributes for influencing native terrestrial reptile and small mammal species assemblages living in remnant forest fragments within Brisbane, south-east

Queensland. The first objective was to review the contemporary Australian urban ecology literature in order to determine the current knowledge base and identify priority areas for future research (Chapter 2). This literature review identified five major gaps in the current urban ecology knowledge base: (1) a lack of multiple species studies; (2) a bias in focal species choice towards more easily identified avian assemblages, or iconic/emotive mammals such as koalas and gliders; (3) a lack of spatially-explicit studies that specifically investigate influences across a hierarchy of multiple spatial levels; (4) very few long-term studies that consider temporal variations; and, (5) few attempts by ecologists to integrate outcomes into urban planning and management. By identifying these gaps, this literature review has provided an essential foundation for designing future projects with priority to filling these knowledge gaps. This project was designed to address these gaps, with the exception of temporal variations (knowledge gap 4), which was not possible within the short time-frame of a thesis.

The second project objective was to investigate the success and cost efficiency of combinations of survey methods. This was achieved by comparing the detection success and cost-efficiency of the six detection methods employed during fauna surveys. In Chapter 3, I presented the findings of these comparative analyses and made recommendations for method selection in future studies investigating similar species assemblages. Reptiles were best surveyed using a combination of pit-fall traps and direct observations, whereas the most optimal survey methods for detecting small and medium sized mammals were cage and Elliott traps, coupled with hair funnels. Such knowledge is important for maximising fauna survey success and minimising the many thousands of dollars spent each year, by researchers and environmental consultants, on terrestrial fauna surveys.

The third objective was to determine the relative importance of local-level habitat structure and habitat composition for influencing reptiles and mammals. This objective was achieved by investigating patterns and relationships in the data collected from fauna and habitat field surveys conducted at each survey site. I presented the analyses of, and conclusions drawn from, these patterns and relationships between species assemblages and local-level habitat attributes in Chapter 4, concluding that habitat structure, rather than habitat composition was of most importance for reptile and small mammal species compositions. This information provided important information about the key habitat elements important for prioritising management actions at the local-level. The findings also provide a comparative study for other researchers, particularly for investigating differences in fine scale habitat requirements of various species within urban landscapes, and between differing land-uses.

Objective four investigated the importance of local-level habitat structure and composition relative to patch size and shape, and landscape composition and configuration. This objective was achieved by testing a set of *a priori* models of the importance of multi-level habitat attributes for influencing reptile and small mammal species richness. I applied hierarchical partitioning and model averaging to the local-level fauna and habitat survey data, as well as patch and landscape-level measurements obtained from GIS data and high resolution Quickbird satellite imagery. These analyses and findings were presented in Chapter 5. Reptiles and mammals were found to be influenced by habitat attributes occurring at more than one spatial level. Landscape composition and configuration, and local-level habitat structure and composition were of most importance for reptiles. Mammals responded most to landscape composition, followed by local-level habitat structure and composition, with patch size and shape being of lowest relative importance. The specific attributes of importance within each spatial level varied between reptile and mammal species assemblages. There was also some variation in important habitat attributes between the single level (local-level) and multi-level analyses. For instance, the number of termite mounds was a key habitat attribute when only the local-level was considered, yet this variable was not of key importance when influences at multiple spatial levels were examined. These findings provide information on important habitat and landscape features that should be given priority attention by urban planners and managers. The findings also highlight the need for a multi-scaled, multi-species perspective to be adopted by researchers when designing and conducting research projects, and by urban planners and managers when planning for future development or managing current and future landscapes.

The final project objective was to translate these multi-level key ecological attributes into a set of guidelines for informing urban planning and management decisions. These guidelines and a decision-support tree were presented in Chapter 6, where I identified how planning frameworks and local management strategies at local to State levels may integrate the key ecological attributes into practical actions. Examples were provided based on the Brisbane case study. I also urged ecologists and decision-makers to form cooperative partnerships and maintain an open dialogue in order to promote urban development and management that is underpinned by sound, scientific knowledge. Further, I recommended more cross-boundary planning and management to facilitate biodiversity conservation across landscapes, rather than independent actions within different boundaries that result in a mismatch between ecosystem boundaries and human-imposed boundaries (Saunders and Briggs 2002). The recommendations presented in Chapter 6 form a crucial step in integrating ecological knowledge into practical application, and represent a method for ecologists to convey priority habitat elements and requirements for different species in a way that may be easily understood and integrated in decision-making processes.

7.2. BROAD CONTRIBUTION TO ECOLOGICAL THEORY

This thesis addressed the complexities of planning and managing urban landscapes for the effective conservation of multiple native fauna species. The findings have contributed to broad ecological theory by extending the current knowledge base to include the habitat requirements of small ground-dwelling vertebrates in urban environments. In addition, the findings have advanced urban ecology knowledge by presenting a method for determining the relative importance of multi-scaled habitat requirements of multiple species. To the best of my knowledge, the project's investigation of survey method success and cost-efficiency is the first of its kind. Further, I believe the investigation and prioritisation of multi-scaled environmental elements for terrestrial reptiles and small mammals in an urban landscape, is the first of its kind in Australia, and only one of a handful of similar studies in the world. This study has demonstrated the importance of explicitly considering multiple species and multiple spatial levels. The three major conclusions drawn about the habitat requirements of terrestrial reptiles and small mammals in an urban landscape support the broad consensus from many studies on other fauna groups (mainly birds) in, usually, non-urban habitats (e.g., Andrén 1994; Bentley and Catterall 1997; Forys and Humphrey 1999; Lindenmayer et al. 1999; Howell et al. 2000; Brotons et al. 2003; Krauss et al. 2003; Riffell et al. 2003; Kanowski et al. 2006):

i) *A landscape perspective is vital for effectively conserving biodiversity in urban landscapes (Chapters 5 & 6).* The findings from the Brisbane case study identified landscape composition and configuration elements as most important for influencing reptile and small mammal species richness within remnant fragments. This means that research studies that only consider elements of the habitat patch (or other single level), may mask or exaggerate the importance of elements across multiple spatial levels. Furthermore, conservation actions that focus only on the habitat patch itself will be unsuccessful for effectively conserving viable species populations for the long-term. It is therefore vital that landscape elements, such as the amount of suitable habitat, the degree of connectivity, and the matrix, are explicitly considered by urban researchers, planners, and managers.

ii) *Structural complexity is more important than floristic diversity (Chapter 4).* For terrestrial reptiles and mammals, maintaining high vertical and horizontal structural complexity within habitat patches was more important than the floristic composition of patches. This finding is particularly important for guiding common urban habitat management actions, such as revegetation and rehabilitation, for which the first step is often aesthetic clearing of natural debris, and the removal of human litter and non-native plants. However, in certain cases, these actions may decrease structural complexity, which will negatively influence the occurrence and diversity of reptiles and small mammals. For instance, natural debris such as fallen wood, human litter such as old car bodies and tin sheets, and certain weeds, may actually be fulfilling important structural requirements. It is therefore important that the structural complexity of focal habitats is assessed and steps are taken to maintain or increase structural elements.

iii) *Landscapes must be managed for multiple species (Chapters 2-6)*. The Brisbane case study showed that the relative importance of habitat attributes varied between reptiles and small mammals. When compared to the scientific literature, these important attributes were also found to vary in comparison to other fauna species and habitat types. This indicates that research focussed only on a single species may obscure or exaggerate the relative importance of habitat variables for multiple species. As a result, urban planning and management decisions that are based on recommendations from single-species research are unlikely to be successful in conserving biodiversity. Researchers, urban planners, and managers must, therefore, take steps to: (a) recognise the full complement of species present within the landscape; (b) understand the multi-level habitat requirements for these species; and, (c) plan and manage urban development based on the habitat requirements of multiple species.

This study also provided new knowledge on the success and cost-efficiency of combinations of different trapping methods. Although tangential to the main project aim, this new knowledge provides an important comparative study to help refine the selection of appropriate survey methods, and manage commonly limiting resources (e.g., time and money), in future terrestrial fauna surveys. The analyses presented (Chapter 3) also provide an example for analysing the success and efficiency of survey methods in future studies.

I additionally showed how research findings may be integrated into regional and local planning schemes and local management strategies (Chapter 6). The translation of research outcomes into a format readily understood and applied by urban planners and managers is a vital step to bridging the gap between researchers and decision-makers. Without the integration of urban ecology research findings, urban planners and managers are ill-equipped with the knowledge necessary to make decisions regarding priority actions, and funding/resource allocation, for successfully achieve biodiversity conservation within the urban context.

7.3. IMPLICATIONS FOR URBAN PLANNING AND MANAGEMENT

This findings presented in this thesis have significant implications for future urban planning, and current and future urban management. Of high priority is the need to plan and manage urban landscapes based on the requirements of multiple species. If the requirements of only a single species or a small subset of species are considered, planning and management actions are likely to be ineffective for conserving a wide range of species. Complementing this, is the need for decision-makers to adopt a hierarchical landscape perspective, which considers environmental attributes across the nested hierarchy of spatial levels, rather than just considering attributes within a single spatial level (e.g., patch-level). This perspective also promotes the landscape as a heterogeneous mosaic of land-use types, rather than a binary landscape of habitat and non-habitat. Accordingly, it is important to plan and manage the matrix between habitat fragments, as well as the forest fragments themselves. To effectively achieve this landscape perspective and recognise biodiversity conservation across urbanising landscapes requires inter-disciplinary and cross-boundary collaboration and communication.

7.4. FUTURE RESEARCH

One of the major limitations to this project is that it occurred over a short time frame. Many of the species investigated in the project, particularly the small mammals, exhibit distinct seasonal fluctuations in populations, which, coupled with their cryptic behaviour, are likely to have resulted in a proportion of false-absence records. That is, sites that were found to be devoid of certain species may actually have been occupied, but the species were not detected during the survey period. To minimise the influence of false-absence records, we repeat surveyed sites in two separate years during the spring/summer breeding seasons; the time of year when species were most active, and so were most likely to be detected. In addition, we surveyed as many sites as possible within the short time frame. We recognise that additional sites, longer survey periods, and additional repetitions over more years and seasons are likely to have strengthened our detection records. In addition, short-term 'snap-shot' surveys, such as that conducted for this project, are unable to determine the status (i.e. stable, increasing, or declining) and dynamics (i.e. extinction and colonisation rates) of species populations. For instance, would the same results be found in another five years and/or with different rainfall history? These are important questions for future research, which should attempt to conduct fauna surveys over longer time frames and, where possible, at more sites. This will enable a more comprehensive investigation of the ecological conditions that favour spatial population dynamics, which will provide important information for influencing planning and management decisions.

The selection of effective combinations of survey methods (Chapter 3) also helps to minimise false-absence records by enabling the most effective combinations to be selected and used intensively during site surveys. It would be useful, therefore, for future studies to also include success and efficiency calculations in their results. Further, a study that extends this investigation by applying a more structured and stratified approach to answering the question of how many, and what kind of traps are best, would be a valuable contribution to the ecological knowledge base.

7.5. CONCLUSION

Biodiversity conservation is a significant and complicated challenge in urban and urbanising landscapes. However, it is necessary if we are serious about conserving viable populations of a range of native fauna in the long-term. Researchers, decision-makers, and community factions alike will need to change their perspective and work together to ensure conservation across multiple spatial levels takes priority in the increasing demand for space. This will require a comprehensive understanding of the ecology of multiple species to inform decision-making processes. The findings and guidelines presented in this thesis provide important information for conserving reptiles and small mammals in Brisbane's lowland forest remnants, as well as a comparative basis for future research, and a scientific framework for guiding decision-making processes in other urban areas.

(a) Storr's rainbow skink (*Carlia vivax*)
- Family: Scincidae
- Active, sun-loving skink.
- Average size: SVL 47 mm.
- Distribution: Queensland, New South Wales.
- In Brisbane: restricted to suitable bushland habitat in peri-urban landscapes.

This photo shows me holding a breeding male Storr's rainbow skink.

(b) Eastern water skink (*Eulamprus quoyii*)
- Family: Scincidae
- Sun-loving skink.
- Average size: SVL 115 mm.
- Distribution: Queensland, New South Wales, South Australia.
- In Brisbane: found along waterways where suitable cover is available, including inner suburbs.

Photo from: http://www.users.bigpond.com/neen105/Lizard3.htm

(c) Secretive skink (*Lampropholis amicula*)
- Family: Scincidae
- Secretive, sun-loving skink.
- Average size: SVL 35 mm.
- Distribution: South-east Queensland, north-east New South Wales.
- In Brisbane: restricted to suitable remnant bushland habitat in peri-urban landscapes.

This photo shows a secretive skink on my hand.

(d) Garden skink (*Lampropholis delicata*)
- Family: Scincidae
- Sun-loving skink.
- Average size: SVL 51 mm
- Distribution: Queensland, New South Wales, Victoria, Tasmania, South Australia.
- In Brisbane: Ubiquitous, thrives in disturbed areas such as suburban gardens.

This photo shows an adult garden skink on my hand.

Plate 8. Skinks: (a) Storr's rainbow skink; (b) Eastern water skink; (c) Secretive skink; (d) Garden skink.

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