## Biodiversity of Urban Riparian Systems: Application of Ecological Data to the Design of Environmental Planning Instruments



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Cover image: Riparian vegetation within Maddison Reserve, Pymble Ku-ring-gai Local Government Area, Sydney.

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## Summary

Protecting minimum riparian corridor widths is a common approach by government authorities seeking to mitigate impacts of urban development on riparian and aquatic ecosystems. However, little data exists to verify the efficacy of these policies for biodiversity conservation. This thesis presents research to this end thereby contributing to urban environmental planning practice.

To promote ecological sustainability, scientific concepts of biodiversity and urban river function need to be connected closely with environmental policy. Following discussion of these themes, the relationship between biodiversity and corridor width was assessed by comparing ant and plant communities from 18 riparian corridors in the Ku-ring-gai Local Government Area, Sydney. Corridor width was not related to ant and plant species diversity but was related to the assemblage composition of both groups. However, substantial data variability remained unexplained.

This variability was investigated through exploring relationships between biological and environmental data. Reserve perimeter:area ratio and the gradient of the hillslope were identified as significant predictors of ant and plant richness. Corridor width was the best predictor of plant composition overall due to increased habitat heterogeneity in wider corridors, while soil pH was the best predictor of streamside plant communities. Changes in plant composition were driven primarily by exotic plants. Ant composition was influenced by the proportion of adjacent road cover, vegetation type, and reserve connectivity.

In response to these results the affect of land use on riparian ant communities was assessed through comparing urban and rural catchments in north-west Sydney. Surprisingly, no significant differences in ant diversity or composition were found, but ants responded the dominance of weeds within the riparian zone.

This work highlights the complexity of biodiversity patterns in urban riparian systems and suggests that in addition to setting minimum corridor widths, managers should consider local and catchment-wide activities and promote active management of the riparian zone. I, Christopher David Ives, certify that this thesis has not been submitted for a higher degree to any other university or institution. The work presented in this thesis is my own and was undertaken during the course of my PhD candidature. Except where acknowledged, this thesis is comprised entirely of my own work.

Signed,

Christopher D. Ives 15<sup>th</sup> August 2011

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Dedicated to the memory of Maurice Ives

# CHAPTER 1

## **General Introduction**

"Ultimately, BMP's (Best Management Practices) are likely to be chosen on the basis of their *demonstrated effectiveness* in a particular landform and human setting and of how much *society values* the expected benefit to the stream ecosystem"

J. D. Allan (2004)

## 1. Interactions between environmental policy and science

Effective environmental policy is invariably grounded in a scientific understanding of natural systems and processes and how they respond human actions. However, the scientists who pursue knowledge of natural systems are commonly disconnected from the policy makers and land managers who utilise this information to manage them (McNie 2007). This is especially so in local councils and related government agencies where there is a great need for high quality scientific information about ecosystem structure and function at the local scale (Theobald et al. 2000) yet there is a paucity of expertise and resources needed to obtain and interpret this information (Kelly 1998; Miller et al. 2009).

Recently there have been calls for greater interaction between science and policy in order to more effectively achieve ecological sustainability outcomes (Cortner 2000; Cullen 2006; Smythe et al. 1996). This thesis responds to this call. It brings together scientific concepts, theories and methods with policies and management recommendations relevant to the specific demands of local government authorities and those managing biodiversity in urban systems.

## 2. Impacts of urbanisation on biodiversity and ecosystem function

Urban environments are particularly in need of effective environmental and land use policies. They contain valuable ecological assets that are subject to increasing threats from a broad suite of anthropogenic pressures (Beatley 2000; Grimm et al. 2008; McKinney 2002). Despite the density of human populations, cities can contain large numbers of species as a result of high habitat heterogeneity (McKinney 2008; Pautasso 2007; Rebele 1994), and remain the home to many threatened and endangered species (Niemela 1999). Furthermore, the 'green' areas located within the metropolitan boundary are increasingly becoming the primary interaction that many people have with the natural world for much of their life (Miller 2005; Savard et al. 2000). With the number and density of cities continuing to increase globally (Grimm et al. 2008), ecological threats such as habitat fragmentation, biotic homogenisation, pollution, physical disturbance, hydrological modification and the spread of invasive species are placing great stress on the ecosystems and biological communities within urban areas (Grimm et al. 2008; Groffman et al. 2003; McKinney 2006; Newman 2006; Paul and Meyer 2001). The sustainable management of urban environments is therefore one of the great challenges of the 21<sup>st</sup> Century. Ensuring the health and protection of urban river and riparian systems will be a significant part of achieving these outcomes.

### 3. The value of urban riparian corridors

Riparian zones have been defined simply as "the interfaces between terrestrial and aquatic ecosystems" (Gregory et al. 1991). As ecotones, they experience a unique set of physical and ecological processes such as flooding disturbance, sediment deposition, nutrient input, fluctuation of soil moisture, and vegetation succession, which vary continually over time and space (Gregory et al. 1991; Malanson 1993; Naiman and Decamps 1997; Naiman et al. 2005). These dynamic processes generate complex mosaics of interacting assemblages of species (Gregory et al. 1991; Naiman and Decamps 1997; Naiman et al. 2005). Within many cities, linear strips of vegetated riparian corridors represent the last tracts of remnant bushland and are thus highly valuable ecologically, environmentally, socially, economically, and aesthetically (Findlay and Taylor 2006). Physical benefits of vegetated riparian corridors include the improvement of water quality through buffering the stream from nonpoint source sediments and pollutants (e.g. nutrients and pesticides) (Davies and Nelson 1994; Dosskey et al. 2010; Gregory et al. 1991; Matteo et al. 2006; Meador and Goldstein 2003; Meynendonckx et al. 2006; Peterjohn and Correll 1984; Schiff and Benoit 2007; Tufford et al. 1998), mediation of stream temperatures (Beschta 1997; DeWalle 2010), and geomorphic stabilisation of stream bed and banks (McBride and Booth 2005; Schiff and Benoit 2007; Sweeney et al. 2004). Ecological benefits include providing input of energy and organic matter (Benson and Pearson 1993; Keller and MacDonald 1995; Tabacchi et al. 1998), maintaining biodiversity both locally and regionally (Naiman et al. 1993; Pollock et al. 1998; Sabo et al. 2005; Ward et al. 2002), enabling the movement of organisms throughout the landscape

(Bennett 2003; Fitzgibbon et al. 2007; Forman 1995; Johansson et al. 1996; Naiman and Decamps 1997; Rosenberg et al. 1997), providing terrestrial habitat for a range of fauna (Crawford and Semlitsch 2007; Lees and Peres 2008; Pennington et al. 2008; Steinberger 1996), and promoting the ecological integrity of aquatic ecosystems (Horwitz et al. 2008; Meador and Goldstein 2003; Miltner et al. 2004; Rios and Bailey 2006; Roy et al. 2007; Schiff and Benoit 2007; Snyder et al. 2003) In addition to these services, riparian corridors in urban landscapes often provide significant aesthetic, recreational and psychological benefits to city residents (Ahern 1995; Kenwick et al. 2009; Loomis et al. 2000; Miller 2005).

## 4. Urban riparian corridor management

Riparian corridors have traditionally been retained within urban landscapes because of difficulty in mitigating flooding risk in these areas and problems associated with development in close proximity to streams. However, these systems are becoming increasingly threatened by urbanisation, both in terms of outright clearing of vegetation to make way for new development and the physio-chemical impacts related to catchment scale processes. These impacts include the modification of hydrological regimes, input of nutrients, increased contamination by pollutants and the spread of invasive species (Groffman et al. 2003; McKinney 2002; Paul and Meyer 2001; Walsh et al. 2005). In recent years, greater recognition by local government authorities of these threats and the environmental and social values of riparian corridors has led to their protection in environmental policies and statutes (UDIA 2009; Wenger and Fowler 2000).

One of the primary parameters typically incorporated into riparian protection policies is the minimum width of the riparian buffer or setback. This is because planning ordinances rely regularly on consistent guidelines that can be applied at the lot-scale on a case-by-case basis, and because many of the physical benefits of riparian buffers (e.g. temperature, sediment filtration, shading etc.) can be predicted to a reasonable level of accuracy as a function of buffer width (Beschta and Weatherred 1984; Castelle et al. 1994 and references therein; DeWalle 2010). However, with the legislative ascent of biodiversity protection goals (e.g. Environmental Protection and Biodiversity Conservation Act 1999 (Cth)) and the rise of holistic ecosystem management principles (Christensen et al. 1996), the importance of riparian corridors for protecting biodiversity has achieved greater National State and local priority. One of principal challenges facing these policies is their validation and scientific justification. Indeed, this challenge was articulated in a judgement by the NSW Land and Environment Court where McClellan CJ held that the weight to be given to any planning policy depended in part upon "the extent, if any, of research and public consultation undertaken when creating the policy" (Stockland Development Pty Ltd v Manly Council [2004] NSWLEC 472 at [92]).

A key difficulty in the application of science to policies that seek to conserve biodiversity is the inherent complexity of biodiversity, both as a concept and in terms of the enormous variability expressed by biological communities over time and space and in response to heterogeneous environmental factors. Although a number of studies have sought to identify specific corridor width guidelines for the protection of riparian biodiversity (e.g. Barton et al. 1985; Cross 1985; Darveau et al. 1995;

Frimpong et al. 2005; Keller et al. 1993; Lees and Peres 2008; Semlitsch and Bodie 2003; Spackman and Hughes 1995), the results are usually only applicable to the specific environment and taxon under investigation (see Figure 1). Consequentially, there is a need for research undertaken in local systems that can feed directly into the requirements of policy designed to protect them.



**Figure 1:** The range of riparian corridor widths required by various taxa as found by a selection of studies. Dashed lines represent the upper limits of the minimum widths identified in the research. Note the significant discrepancy of values both within and between taxonomic groups.

## 5. The Ku-ring-gai Council Riparian Policy

In 2004, Ku-ring-gai Council in Sydney's north introduced a riparian policy that responded to the need to provide structured support for the protection of rivers and streamside habitats from the pressures unique to the region. Like many riparian management frameworks, the Ku-ring-gai Policy (Ku-ring-gai Council 2004) was based upon minimum corridor widths from which human activity (such as vegetation removal or development) was to be excluded. However, unlike some policies, the widths prescribed varied according to the environmental condition and significance of the stream/riparian zone and an associated set of objectives. All reaches within the stream network were assigned to one of the three categories presented in Table 1.

**Table 1** Riparian corridor categories and respective environmental objectives fromKu-ring-gai Council's Riparian Policy (Ku-ring-gai Council 2004).

Category	Minimum Corridor Width	Primary Objectives	Description
<i>Category 1</i> : Environmental Corridor	40 m	Provide landscape linkages for wildlife + Category 2 and 3 objectives	Contains most ecologically valuable sites in municipality.
<i>Category 2</i> : Terrestrial and Aquatic Habitat	20 m	Enhance habitat and preserve natural geomorphic and biological features.	Permits natural geomorphic processes to operate; limited wildlife movement.
<i>Category 3</i> : Bed and Bank Stability	10 m	Filter runoff and stabilise geomorphic features.	Limited habitat value but important for overall catchment health. Streams often heavily modified.

These categories were based upon research by the then NSW Department of Infrastructure, Planning and Natural Resources (DIPNR) (2004), which borrowed heavily from research in the USA on urban riparian form and ecological function along with two earlier commissions of inquiry into the long-term planning of regions south of Sydney (Cleland and Carleton 2003; Simpson 1999). Given the fact that similar riparian management policies are set to be applied to other local government jurisdictions in Sydney including newly designated urban growth regions in the northwest and south-west sectors of Sydney (Don Fox Planning 2003), there is a great need for local research on the ecology of these riparian zones if these policies are to be legally upheld and the long term protection of urban riparian zones is to be ensured.

## 6. Thesis rationale

The purpose of this thesis to provide insight into the biodiversity patterns and ecological functioning of urban riparian systems in such a way as to be applicable directly to the management of urban riparian corridors. In doing so, this thesis is positioned at the nexus between science and environmental policy, using the management of streams in the Ku-ring-gai LGA as a local case study. Indeed, results from research presented herein have already being used in the formulation of Ku-ring-gai Council's Development Control Plan 47 (Water Management) (Ku-ring-gai Council 2005) and Local Environmental Plan (Ku-ring-gai Council 2010). However, this thesis is not narrowly constrained to issues specific to the Ku-ring-gai LGA. It also delves into the wider philosophical issues of biodiversity protection, examines closely the practical challenges of local environmental policy formulation and implementation, and contributes to the discipline of urban ecology through investigation the response of riparian biological communities to urbanisation. The specific aims of this thesis are as follows:

1) To explore the role of science in the formulation and implementation of local environmental policies, and to provide a framework for the integration of biodiversity concepts into such policies;

2) To investigate the relationship between riparian corridor width and biodiversity in an urban setting;

3) To identify the principal environmental factors affecting riparian biodiversity; and

4) To synthesise these findings in the context of national and international urban ecological research and provide recommendations to the formulation of local environmental policies.

In addressing each of the aims of the thesis, the following chapters draw from various disciplines of environmental law, urban planning, applied ecology and biodiversity informatics, each having their own nuances in terminology and understanding of terms. While there is a potential risk that certain readers with experience in one field may be unfamiliar with the content of chapters that focus heavily on another, for the purpose of a research document it was considered prudent that each chapter conform to the style and language anticipated by that discipline. This is particularly the case when a thesis such as this is presented as a coherent collection of research papers. However, in order to maintain consistency and linkage between the chapters of this thesis, each chapter is prefaced with a brief summary of its objectives and connection to the broader aims and structure of the thesis.

### 7. Thesis structure and chapter outlines

**Chapter 2** – Legal and Policy Frameworks for the Conservation of Biodiversity and Riparian Environments in Urban Environments.

This chapter is positioned to outline the context for the management of urban riparian biodiversity in Sydney. Furthermore, after discussing the pressing challenges to achieving ecological sustainability within urban areas, each paper presents an alternative paradigm or set of recommendations, that if adopted would enhance the protection of urban riparian systems into the future

Paper 1: "New directions in urban biodiversity conservation: The role of science and its interaction with local environmental policy".

This paper provides the conceptual foundation for the following data chapters through exploring the relationship between science and local environmental policy in the context of biodiversity conservation. It begins by discussing the breadth of both the scientific concept of biodiversity and techniques for its measurement. It distils these ideas into a framework that can be easily interpreted and applied by local environmental planners and practitioners. The remainder of the paper discusses the mandate local governments have for protecting biodiversity and the challenges commonly faced in seeking to achieve holistic biodiversity conservation within existing legal and political frameworks. The article concludes with a discussion of possible solutions to the challenges previously outlined and a call for greater integration of fine-

scale empirical data on biodiversity into policy. This paper has been published in the *Environmental and Planning Law Journal*:

Ives, C. D., Taylor, M. P., Nipperess, D. A. and Davies, P. 2010. New directions in urban biodiversity conservation: The role of science and its interaction with local environmental policy. *Environmental and Planning Law Journal* **27**(4): 249-271.

Paper 2: "Urban rivers and riparian systems: directions and recommendations for legislators, policy makers, developers and community users"

This paper provides the context for the management of river and riparian systems in Sydney. It explores the historical evolution of laws pertaining to riparian protection, identifies areas of inadequacy within current legal frameworks, and introduces suggestions for how urban rivers and riparian zones can be protected more strongly in the future. In particular, the unique ecological and biodiversity values of urban riparian zones are discussed and proposed as a key argument for a renewal of riparian management approaches. This paper has published in the *Environmental and Planning Law Journal*:

Davies, P.J., Ives, C.D., Findlay, S.J. and Taylor, M.P. 2011. Urban rivers and riparian systems: Directions and recommendations for legislators, policy makers, developers and community users. *Environmental and Planning Law Journal* **28**(5): 313-331.

**Chapter 3** – The relationship between riparian corridor width and biodiversity.

Paper: "The influence of riparian corridor width on ant and plant assemblages in northern Sydney, Australia"

This paper seeks to explore the validity of using corridor width as a primary parameter in riparian policies for the protection of biodiversity. Communities of ants and plants were sampled within 18 corridors of varying widths in the Kuring-gai LGA. These groups were selected because of their high species richness, fine spatial patterning and sensitivity to environmental change. Measures of species diversity, community composition and ecological function were related to corridor width via a range of univariate and multivariate statistical techniques introduced initially in Paper 1 of Chapter 2. Ecological processes that are likely to have given rise to the observed patterns in ant and plant diversity are outlined, before the implications of the results to urban riparian policy are discussed. This paper has been published in the journal *Urban Ecosystems*:

Ives, C. D., Hose, G. C., Nipperess, D. A. and Taylor, M. P. 2011. The influence of riparian corridor width on ant and plant assemblages in northern Sydney, Australia. *Urban Ecosystems* **14**(1): 1-16.

**Chapter 4** – Exploration of the principal factors driving variation in urban riparian biodiversity.

Paper: "Environmental and landscape factors influencing ant and plant diversity in suburban riparian corridors"

This chapter builds upon the findings from Chapter 3 by investigating the source of the variation in ant and plant communities that could not be accounted for by corridor width alone. Whereas the fundamental research question in Chapter 3 was determined by existing policy (i.e. is biodiversity influenced by corridor width?), Chapter 4 takes a broader perspective and seeks to identify the principal determinants of riparian ant and plant biodiversity through considering a suite of environmental variables. In doing so, this paper assesses the legitimacy of corridor width in forming the foundation of many riparian protection policies in the context of the influence of the multitude of other variables. This paper has undergone substantial peer review and has been published in the journal *Landscape and Urban Planning:* 

Ives, C.D., Hose, G.C., Nipperess, D.A. and Taylor, M.P. (2011) Environmental and landscape factors influencing ant and plant diversity in suburban riparian corridors. *Landscape and Urban Planning* **103**: 372-382. **Chapter 5** – The influence of catchment land use on riparian ant diversity.

Paper: "Comparison of riparian ant assemblages between rural and urban catchments"

This chapter presents results from a study in north-west Sydney designed to test specifically the impact of urbanisation at the catchment scale on riparian ant communities. The paper builds upon results from Chapter 4 that identified the surrounding landscape context as being important in determining the diversity of riparian ant assemblages. It tests the influence of land use by contrasting sites selected carefully from urban catchments with those from catchments dominated by rural land use. Similar biodiversity assessment techniques to those presented in the previous two chapters were used to outline differences in ant diversity, assemblage structure and ecological function. The discussion revolves around how the results relate to existing ecological theory and empirical data pertaining to biological responses to land use change. The paper finishes with a set of recommendations to managers, policymakers and planners seeking to promote the ecological integrity of riparian zones in both urban and peri-urban catchments. This chapter has been prepared for submission to the journal *Landscape Ecology*.

### Chapter 6 – Discussion.

This chapter reiterates and synthesises results from the previous data chapters and discusses in greater depth explanations for the relationships observed with regard to those anticipated at the start of the study. Additionally, implications of the results to urban biodiversity management and planning are outlined before postulating future research directions and exploring new and innovative approaches to the conservation of urban riparian systems.

## Chapter 7 – Conclusions

The final component of this thesis summarises the key findings of the research chapters and distils some general conclusions from the thesis as a whole.

## Appendices

Appendix 1 – Paper: "Legislative and policy challenges for the protection of biodiversity and bushland habitats: An evidence-based approach"

Through exploration of Australian case law and scientific research, this paper discusses the importance of the application of empirical science to the management of urban bushland environments. Some similar issues to those presented in Paper 1 of Chapter 2 are presented and thus it is included as an appendix rather than a paper in the main thesis. However, it does provide valuable insight into the workings of urban bushland management in Australia from a legislative and dispute resolution perspective. The paper has been published in the *Environmental and Planning Law Journal*:

Taylor, M. P. and Ives, C. D. 2009. Legislative and policy challenges for the protection of biodiversity and bushland habitats: An evidence-based approach. *Environmental and Planning Law Journal* **26**: 35-48.

Appendix 2 – Discussion of the use of ants and plants as biodiversity surrogates.

Since ants and plants studied in this thesis are assumed to possess some value as biodiversity surrogates, exploration of the reliability of this assumption is important. In this appendix, data collected from previous chapters are used to assess the degree to which biodiversity patterns in one taxonomic group reflect those in the other. The implications of these results to urban biodiversity management are discussed. In addition, the use of the ant functional groups approach to indicating ecological function (Andersen 1995) is assessed critically based upon results derived from Chapters 3 and 5.

#### Appendix 3 – Raw data files

The raw biological and environmental data acquired during this study along with metadata files are included as an electronic appendix at the back of the thesis.

#### Chapter 1 Introduction

The 'thesis by publications' format of this thesis differs significantly from the traditional PhD dissertation format. Stand-alone literature review and method chapters have not been included to avoid repetition. However, these aspects are dealt with in full within the relevant chapters in the body of the thesis. Additionally, as many of the papers have multiple authors, the percentage contribution of each author and the roles each carried out are indicated at the beginning of each chapter. It is worth noting that although multiple authors are listed on the papers, my contribution as first author is not dissimilar to that which might be expected in a traditional thesis. The inclusion of my supervisors as co-authors reflect their contribution to refining the manuscripts for publication and would mimic the degree of input expected of supervisors when assisting a student prepare a traditional thesis for submission.

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# CHAPTER 2

# Legal and Policy Frameworks for the Conservation of Biodiversity and Riparian Environments in Urban Landscapes

The following two papers outline the theoretical, legal and policy context for the protection of urban biodiversity and riparian environments. With the intention of bridging the gap between scientists and policymakers, they were submitted to the *Environmental Planning and Law Journal*, which is Australia's leading publication in its field and is read by lawyers, land-use planners and environmental practitioners.

Paper 1: "New directions in urban biodiversity conservation: The role of science and its interaction with local environmental policy".
CD Ives, MP Taylor, DA Nipperess and P Davies
Published in the Environmental Planning and Law Journal (2010) 27 pp.249-271

This paper seeks to bring together the science and practice of biodiversity management in the context of local environmental policy. It firstly reviews the development of the biodiversity concept in the scientific literature and outlines the historical growth of biodiversity conservation as a feature of legal and policy instruments in Australia. This article argues that the sustainable management of urban ecosystems can only be achieved through application of the scientific understanding of the biodiversity concept to local land use planning decisions, driven by principles of Ecologically Sustainable Development. A new and practical framework for decision-making is outlined that calls for the application and utilisation of scientifically established biodiversity assessment methods in land use planning at a range of scales.

Author Contributions (Ives et al., 2010):

CD lves: 85% Conception of ideas; literature review; writing of text. MP Taylor: 5% Assistance with development of themes; advice on legal context; editing of text. DA Nipperess: 5% Assistance with explanation of scientific methods; editing of text. P Davies: 5% Advice on policy and local environmental planning instruments; editing of text.

Paper 2: "Urban rivers and riparian systems: directions and recommendations for legislators, policy makers, developers and community users"P Davies, CD Ives, S Findlay and MP Taylor

Published in the Environmental Planning and Law Journal (2011) 28 pp. 313-331.

This paper arose from a need identified in 2004 when Ku-ring-gai adopted its Riparian Policy. It describes the evolution of urban river and riparian management in Australia. The article outlines firstly the failure of traditional management approaches and policies to adequately protect the ecological integrity of these systems as a result of Eurocentric and utilitarian perceptions of urban river systems. After proposing new definitions for a "river" and "riparian zone" appropriate to the Australian urban context, the article suggests a number of reforms to river and riparian policy and management approaches to ensure more effective future protection of these systems. This paper paints a picture of the broad legal and policy landscape to which results from following chapters are to be applied.

Author Contributions (Davies et al., 2011):

# P Davies: 60%

Conception of ideas; writing of text.

# **CD lves: 25%**

Development of themes; writing of text, particularly sections pertaining to riparian zones, their definition, function and management; editing.

# S Findlay: 5%

Contribution to ideas and arguments; editing text.

# MP Taylor: 10%

Conception of ideas; oversight of the article, editing text.

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Christopher D Ives , Mark Patrick Taylor , David A Nipperess and Peter Davies (2010) New directions in urban biodiversity conservation: the role of science and its interaction with local environmental policy. *Environmental and Planning Law Journal*, vol. 27, no. 4, pp. 249-271.

Chapter 2 Scientific and Policy Foundations

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Peter J. Davies, Christopher D. Ives, Sophia J. Findlay and Mark P. Taylor (2011) Urban rivers and riparian systems – directions and recommendations for legislators, policy makers, developers and community users. *Environmental and Planning Law Journal*, vol. 28, no. 4, pp. 313-331. Chapter 2 Scientific and Policy Foundations

# CHAPTER 3

# The Relationship Between Riparian Corridor Width and Biodiversity

Paper: "The influence of riparian corridor width on ant and plant assemblages in northern Sydney, Australia"CD Ives, GC Hose, DA Nipperess and MP TaylorPublished in Urban Ecosystems (2011) 14: 1-16

This chapter seeks to identify whether a relationship exists between biodiversity and riparian corridor width, within the Ku-ring-gai Local Government Area. It follows directly on from Paper 1 in Chapter 2 as an example of policy-relevant research, and is designed to provide data that can be used to assess the efficacy of riparian setbacks within local planning controls that aim to protect "biology and riparian habitat" (discussed in Paper 2, Chapter 2). Using ant and plant communities as indicators of biodiversity and ecosystem function, it applies the framework for biodiversity conceptualisation outlined in Chapter 2 (Table 1 of Paper 1). Biodiversity composition, structure and function are assessed using a range of interpretive statistical techniques appropriate at the "habitat" scale. Patterns observed are discussed in the context of ecological theory and recommendations are made for the management of riparian corridors in urban landscapes.

# Author Contributions:

# **CD lves: 85%**

Conception of hypothesis, formulation of methodology, sample collection, data analysis and interpretation, writing of text.

# GC Hose: 5%

Assistance with statistical analysis and interpretation, editing of text.

# DA Nipperess: 5%

Assistance with statistical analysis and interpretation, editing of text.

# MP Taylor: 5%

Oversight of the project, editing of text.

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# CHAPTER 4

# Exploration of the Principal Factors Driving Variation in Urban Riparian Biodiversity

**Paper:** "Environmental and landscape factors influencing ant and plant diversity in suburban riparian corridors"

CD Ives, GC Hose, DA Nipperess and MP Taylor

This paper has been published in Landscape & Urban Planning (2011) 103: 372-382

Although riparian corridor width was found in Chapter 3 to be significantly related to the composition of ants and plants, it did not predict patterns in species richness and a large degree of community variability in both taxonomic groups remained unexplained. This chapter pursues the broader question of "what are the primary drivers of biodiversity patterns in urban riparian corridors?". A range of environmental variables characterising each site and its surrounding landscape were measured and used to model responses in ant and plant diversity. In doing so, it places the relationship between corridor width and biodiversity in the context current and historical urban ecological research on the nature of biological responses to environmental change and thus seeks to identify whether the use of riparian corridor width in planning ordinances is the most effective means of maximising riparian ecological integrity.

# Author Contributions:

# **CD lves: 80%**

Conception of hypothesis; formulation of methodology; collection of biological and environmental data; data analysis and interpretation; writing of text.

# GC Hose: 10%

Assistance with statistical analysis and interpretation, editing of text.

# DA Nipperess: 5%

Assistance with statistical analysis and interpretation, editing of text.

# MP Taylor: 5%

Oversight of the project, editing of text.

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# Environmental and landscape factors influencing ant and plant diversity in suburban riparian corridors

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## ABSTRACT

Urban riparian zones are highly valuable for a range of environmental, social and economic reasons. Despite their ecological complexity, legislative instruments underpinning riparian protection focus predominantly on minimum corridor widths required to fulfill environmental functions. This study sought to identify the dominant environmental variables that influence riparian corridor biodiversity in northern Sydney, Australia. Ant and plant assemblages were sampled from 18 riparian corridors of various widths. Environmental data pertaining to each corridor were collected using field and desktop surveys. Generalised Linear Modelling found perimeter: area ratio and the gradient of the slope perpendicular to the stream to predict significantly ant and plant richness. Plant richness was also influenced by vegeta tion community type and corridor width. When streamside vegetation communities were considered in isolation, many of the same factors were found to influence species richness. Likewise, Distance-based Linear Modelling identified vegetation community type and landscape connectivity as significant predictors of ant composition both in streamside habitats and those located across the full width of each riparian corridor sampled. While width was the best predictor of plant assemblage composition across the whole riparian corridor, streamside vegetation was influenced more significantly by soil pH. This result arose principally due to a dominance of exotic plants in alkaline soils. Although wider corridors were likely to encompass a greater variety of biotic assemblages, landscape and site scale environmental characteristics appeared to be of greater ecological significance. Environmental managers should therefore consider these factors and target weed invasion when seeking to enhance biodiversity in urban riparian corridors.

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### 1. Introduction

Linear strips of vegetation alongside watercourses are common in landscapes heavily modified by human activities (Lee, Smyth, & Boutin, 2004; Naiman, Decamps, & Pollock, 1993; Parker, Head, Chisholm, & Feneley, 2008). These strips extend from the immediate streamside-riparian zone laterally toward the valley margin. They are now retained primarily for the environmental benefits they offer to adjacent stream ecosystems and for their value as remnant bushland within an ecologically degraded matrix but historically they have avoided development because of their steep topography and/or susceptibility to flooding (Osborne & Kovaci, 1993; Tabacchi et al., 1998). These corridors (hereafter referred to as 'riparian corridors', but also sometimes termed riparian buffers,

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setbacks or greenways) are recognised as important ecological conduits, having been used in conservation planning as 'bio-highways' to promote biological connectivity within urban and peri-urban landscapes (Bryant, 2006; Parker et al., 2008). Indeed, from a conservation perspective, riparian corridors can help protect ecological communities well outside the immediate streamside zone often considered to be 'true' riparian habitat.

Conservationists have long appreciated the significant biodiversity values of riparian corridors at local and regional scales (Naiman et al., 1993). Despite this, legislative protection of riparian lands remains narrowly focused on the minimum width of vegetation necessary to minimise the impact of upslope activities on riparian or aquatic ecosystems (Lee et al., 2004; Rodewald & Bakermans, 2006). This focus is common across different countries (e.g. Australia: NSW Water Management Act [2000], USA: Wisconsin DNR, 2007) and land uses (e.g. residential, forestry, agriculture etc.), and is partly a legacy of the well-recognised functions of riparian corridors to trap sediment and pollutants (Dosskey et al., 2010), filter nutrient enriched runoff (Dosskey et al., 2010), stabilise stream channels (Sweeney et al., 2004),

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maintain stream temperatures (Osborne & Kovacic, 1993) and mitigate floods (Weller, Jordan, & Correll, 1998). Indeed, to these ends, specifying minimum riparian buffer widths can be entirely appropriate as there is a good understanding amongst scientists and environmental practitioners of the ways in which streamside vegetation maintains the physical and chemical health of rivers (Dosskey et al., 2010).

The biodiversity of riparian corridors, however, is a product of a complex set of interactions between organisms and their environment across a range of spatial and temporal scales. Biodiversity is therefore difficult to quantify as it can be perceived in many ways and varies greatly over time and space (Magurran, 2004). This inherent complexity means that management approaches that single out one particular characteristic or function of riparian corridors for protection are unlikely to encompass the full range of ecological values they provide (Hess & Fischer, 2001). Although research and conservation approaches have focused predominantly on corridor width (Spackman & Hughes, 1995), riparian biodiversity has been related to a number of other environmental variables such as catchment position (Renofalt, Nilsson, & Jansson, 2005), surrounding land use (Moffatt, McLachlan, & Kenkel, 2004; Rodewald & Bakermans, 2006), soil properties (Désilets & Houle, 2005; Renofalt et al., 2005), and ecological connectivity (Tabacchi et al., 1998; Ward, Tockner, Arscott, & Claret, 2002).

Recent evidence challenges the efficacy of this narrow focus on corridor width to achieve optimal biodiversity outcomes in riparian systems. For example, 94% of explained variation in riparian bird communities in Ohio, USA, was attributed to the degree of urbanisation adjacent to riparian forests, while only 6% was attributed to the width of the riparian corridor (Rodewald & Bakermans, 2006). In the present study, we seek to identify whether site attributes, such as soil type, reserve geometry and topography, and landscape characteristics control the diversity of ants and plants in riparian corridors in the municipality of Ku-ring-gai, Sydney, Australia.

Ants and plants were studied because their fine spatial patterning would enable ecological changes to be measured over small distances such as changes in riparian width. Ants in particular are abundant and biologically diverse at local scales (Pik, Dangerfield, Bramble, Angus, & Nipperess, 2002), respond rapidly to small scale environmental change (Pik et al., 2002), contribute significantly to a range of ecosystem processes (Folgarait, 1998) and can be sampled rapidly with negligible impact on local populations (Ward, New, & Yen, 2001). Likewise, vegetation assemblages are very speciose and used frequently in ecological assessments of environmental impacts (e.g. Moffatt et al., 2004). We explored two key attributes of biodiversity, namely species richness and community composition, to address the following questions:

- (1) Which physical environmental characteristics of riparian corridors and their surrounds influence most greatly ant and plant communities therein?
- (2) Do plant and ant communities respond to similar sets of environmental variables?
- (3) Do ant and plant communities located directly adjacent to the stream respond to environmental factors in a similar manner to communities located further upslope?

Answers to these questions are of significance to both our scientific understanding of ecosystem responses to urbanisation, as well as to policy makers endeavouring to protect urban riparian biodiversity. After discussing our results in regards to ecological pattern and process, we highlight a number of policy recommendations for improved management of urban riparian ecosystems.

### 2. Materials and methods

## 2.1. Study area and sample sites

Ku-ring-gai Local Government Area (LGA) is situated within the northern suburbs of Sydney, Australia and is 8446 ha in size. The geology is comprised of Hawkesbury Sandstone and Wianamatta Group rocks of the Triassic period (~200-250 Mya) that have been eroded deeply along drainage lines, producing steeply incised sandstone valleys. Forty seven percent of the municipality consists of official bushland reserves: 2800 ha of which are in national parks and 1100 ha are in Council managed reserves (Ku-ring-gai Council, 2005). Despite its 'leafy' character, significant urban development has taken place across the LGA since the 1950s in the form of lowdensity housing. In contrast to many other urbanised watersheds, development in Ku-ring-gai has been concentrated in catchment headwaters because of the steep gullies downstream. The result of this trend is a network of incised valleys and associated vegetated corridors located within a suburban matrix. As with many urban environments globally, increasing population pressure is contributing to the encroachment of development into undeveloped bushland, including riparian corridors. These corridors are therefore of great conservation importance in the Ku-ring-gai LGA as they are some of the final remnants of intact bushland. Furthermore, small reductions in corridor width (i.e. <10 m) often create significant economic and political conflict as a result of development pressure stemming from high land values, and the pro-conservation approach of the local government riparian policy (Ku-ring-gai Council, 2004).

In this paper, we use the term 'riparian corridor' in the broadest sense, referring to the lateral extent of bushland that adjoins a watercourse. We acknowledge, however, that the term 'riparian' is often interpreted in different ways by different disciplines and our definition of a riparian corridor may include areas well beyond the immediate river bank that experience little direct influence from the stream. Indeed, our definition of 'riparian' is more in keeping with the use of the term in a planning sense. It is of great interest to study the diversity of the full width of a riparian corridor since the upland component, while possibly not 'riparian' in a strict ecological sense, contributes greatly to the biodiversity of a landscape and are highly valuable in terms of both conservation and land value.

The biodiversity of riparian corridors was characterised by sampling the invertebrate fauna and vascular plants within 18 sites within the LGA (Fig. 1). Sites were of varying riparian corridor widths (the distance between the stream and adjoining human land use) ranging between 3 m and 546 m. Five of these were selected as reference sites where the urban boundary was greater than 125 m from the stream bank. Sites were selected only if they contained similar vegetated corridors on both left and right stream banks.

### 2.2. Data collection

Terrestrial invertebrates were sampled using pitfall traps (diameter 67 mm) containing ethanol and water (ratio 70:30). Traps were buried flush with the ground and were distributed in a series of longitudinal transects that run parallel to the stream channel. Along each transect, three pitfalls were positioned at 15 m intervals to minimise spatial autocorrelation and trap depletion (Sanderson, Rushton, Cherril, & Byrne, 1995). Transects were replicated every 15 m upslope of the streambank. This ensured that sampling effort was proportional to corridor width, and ensured the full characterisation of each site. Sampling was performed in autumn 2006. Invertebrate specimens were identified to Order, and ants (Hymenoptera: Formicidae) were further identified to Genus before being split into morphospecies.



Fig. 1. Map displaying the location of study sites within the Ku-ring-gai Local Government Area.

Vegetation composition alongside the invertebrate samples was surveyed using  $30 \text{ m} \times 10 \text{ m}$  transects positioned around each set of three pitfall traps (Fig. 2). All species of vascular plants were recorded and assigned a cover abundance class between 1 and 8 using the Braun–Blanquet scale (1 = one/few individuals & <5% cover; 2 = uncommon & <5% cover; 3 = common & <5% cover; 4 = very abundant & <5% cover; 5 = 5–20% cover; 6 = 20–50% cover; 7 = 50–75% cover; 8 = 75–100% cover). Surveys were undertaken in autumn and winter 2008.

Environmental data were gathered in the field, remotely via GIS, and through communication with local government personnel. Soil pH at each pitfall trap was tested on site using a field test kit. Orthophotos were analysed in ESRI® ArcGIS (version 9.2) and the polygon measurement tool was used to calculate the area and perimeter of the contiguous area of bushland within which



Fig. 2. Arrangement of pitfall traps (black circles) and vegetation transects (overlaid rectangles) within a site.

each sample site was located. Connectivity was determined using an ordinal scale between 1 (poorly connected) and 5 (well connected) based upon the distance between and density of vegetation linking the study site and nearby areas of significant bushland. The soil landscape classification (Chapman & Murphy, 1989) for each site was also recorded, along with the stream order of the reach (Strahler, 1952). Classification of the native vegetation community as per the Conservation Atlas of Plant Communities in Australia (Specht, Specht, Whelan, & Hegarty, 1995) was recorded to represent the type of biodiversity information usually provided to planners. We note that this classification does not reflect the fine-scale floristic variability actually occurring at a site and was included to test the correlations between mapping units used for decision making and empirical, site-scale data. The presence and nature of bush regeneration activities occurring on a site were ascertained from discussion with environmental managers at Kuring-gai Council. Sites were then allocated a score between 0 and 5 for both the quality and quantity of restoration works. High quality restoration was that which had been undertaken by experienced professionals in an ecologically sensitive manner with demonstrable positive impacts on floral composition and structure. The quantity of restoration refers to the physical extent and intensity of the works that have been undertaken.

The landscape context surrounding each study site was quantified from aerial photographs following the technique employed by Sinclair, Hess, Moorman, and Mason (2005). For both the left and right banks, a 4000 m<sup>2</sup> polygon of grid points spaced 20 m apart was laid over the orthophoto of each site. Points were tallied as intersecting trees/bushland, building, lawn, road, water, pavement or bare earth (Fig. 3).

### 2.3. Data analysis

To identify the dominant factors controlling the biodiversity of riparian corridors, we applied both univariate and multivariate sta-



Fig. 3. Aerial photo of a study site with the grid of points used to quantify adjacent land use.

tistical techniques. We addressed two aspects of biodiversity in our analyses: species richness and community composition. Analysis of vascular plant species richness was performed firstly for all observed species, and secondly for native species only. This enabled any effects of invasive species to be assessed independently of compositional change in native assemblages.

Because biological communities commonly vary with increasing distance upslope from the stream bank, collected ant and plant data were arranged into two datasets that were analysed separately: (i) full-width data consisting of samples from all transects irrespective of their distance from the stream, and (ii) streamside data consisting only of transects located directly adjacent to the stream. Assessing data in this way enabled the effect of hillslope variation on ant and plant assemblages to be isolated and allowed any differences their response to environmental factors according to hillslope position to be identified.

By sampling in proportion to corridor width, sampling effort (i.e. number of vegetation transects or invertebrate pitfalls) was greater in wider corridors. This resulted in more individuals sampled at these sites, which in turn increased the likelihood of encountering more species (see Hill, Curran, & Foody, 1994). Therefore, prior to performing analyses on full-width biological data, adjustments were made to account for this unequal sampling, thus permitting direct comparisons between corridors of different widths. Details of this adjustment are outlined below.

Ordinal or categorical environmental predictor variables were translated into numerical values applicable for statistical analysis. Variables with skewed or unevenly distributed data were transformed where appropriate (see Table 1) to ensure each variable maintained heterogeneity of variance across samples. Pearson correlation tests between environmental variables were performed to identify any that were closely related and thus unnecessary for explaining observed variance in the biological data. Finally, data were normalised to a common standard deviation distribution prior to analysis, to standardise their units of measurement and enable equal contribution of each variable to the data matrix.

### 2.3.1. Richness

Generalised linear modelling was used to test the predictive strength of environmental variables on the species richness of a site. Sampling effort was expected to be highly correlated with site richness and was therefore adjusted for the full-width dataset by using the average transect species richness as the response variable. The variables entered into the model are listed in Table 1, 'Reserve Area' and 'Reserve Perimeter' were highly correlated, therefore 'Reserve Perimeter' was excluded from subsequent model selection. Preliminary linear regression analyses were performed between individual environmental variables and the species richness of ants and plants to identify the strongest independent associations. Stepwise model selection was subsequently used to identify the optimum combination of variables and was guided by the Akaike Information Criterion (AIC) measure of fit. Given a sample size of 18, we chose to limit the number of predictive environmental variables to a maximum of five, to avoid over-fitting the data and providing a more simple, more readily interpretable model (MacNally, 2000).

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### Table 1

P values of single term linear models between species richness and environmental variables.

Predictor variable	Transformation/ units	Full wid	th					Streams	ide				
		Ants		All vege	tation	Native ve	getation	Ants		All vege	tation	Native v	egetation
		t	Р	t	Р	t	Р	t	Р	t	Р	t	Р
Aspect EW	Dummy variable	-1.11	0.29	-0.89	0.39	0.61	0.55	-0.83	0.42	-1.72	0.11	-0.58	0.57
Aspect NS	Dummy variable	1.58	0.13	0.82	0.42	-0.14	0.89	0.56	0.59	1.45	0.17	0.67	0.52
Connectivity	Rank (1-4)	-0.60	0.55	-1.82	0.09	2.29	0.36	-1.17	0.26	-1.75	0.10	0.82	0.43
Gradient	Degrees	-2.85	0.01**	-2.13	0.05	-0.182	0.86	-2.87	0.01**	-1.75	0.10	-0.56	0.58
Area	Logarithm	0.19	0.85	-1.09	0.29	3.43	<0.01**	-0.29	0.78	-0.71	0.49	2.01	0.06*
Corridor width	Logarithm	-0.59	0.56	-1.64	0.12	3.75	<0.01**	-0.73	0.48	-1.34	0.20	0.82	0.43
Perimeter	Logarithm	0.27	0.79	-0.96	0.35	2.92	0.01**	-0.19	0.86	-0.63	0.54	1.73	0.10*
Perimeter:area ratio	-	0.90	0.38	2.28	0.04**	-3.04	<0.01**	1.36	0.19	1.48	0.16	-1.65	0.12
pH		0.34	0.74	0.07	0.95	-3.88	< 0.01**	0.30	0.77	0.87	0.40	-1.70	0.11
Regeneration quality	Rank 1-5	1.56	0.14	0.68	0.51	0.64	0.53	0.99	0.34	0.22	0.83	0.65	0.53
Regeneration time	Years	1.93	$0.07^{*}$	0.25	0.80	0.42	0.68	1.31	0.21	0.17	0.87	0.58	0.57
Soil type	4 factors	2.03 <sup>a</sup>	0.16	1.16 <sup>a</sup>	0.36	0.35ª	0.79	1.74 <sup>a</sup>	0.21	2.66ª	0.09*	0.25 <sup>a</sup>	0.86
Stream order	1-5	-0.42	0.68	-1.19	0.25	1.83	0.09*	0.03	0.98	-1.13	0.28	0.49	0.63
Vegetation community type	6 factors	0.29 <sup>a</sup>	0.90	2.40 <sup>a</sup>	0.10*	0.64 <sup>a</sup>	0.68	0.25 <sup>a</sup>	0.96	1.86 <sup>a</sup>	1.18	0.71 <sup>a</sup>	0.63
% Adjacent bare earth	Logarithm	0.61	0.55	-0.75	0.46	0.80	0.44	0.24	0.81	0.46	0.65	1.30	0.21
% Adjacent buildings		-0.20	0.85	0.86	0.40	-3.02	<0.01**	0.25	0.81	0.27	0.79	-2.01	0.06*
% Adjacent grass		0.29	0.78	0.91	0.38	-0.55	0.59	0.05	0.96	1.60	0.13	0.66	0.52
% Adjacent paved		-0.14	0.89	1.46	0.16	-2.14	0.05**	0.32	0.76	1.80	0.09*	-0.54	0.60
% Adjacent road		-1.17	0.26	0.37	0.71	-2.95	< 0.01**	-0.60	0.56	0.24	0.81	-1.56	0.14
% Adjacent tree		0.23	0.83	-0.90	0.38	2.53	0.02**	-0.04	0.97	-1.03	0.32	0.95	0.35
% Adjacent water		-0.33	0.75	-0.16	0.87	-2.16	0.05**	0.08	0.94	-0.38	0.71	-1.82	0.09*

\* Significant at  $\alpha = \le 0.1$ .

<sup>\*\*</sup> Significant at α = ≤0.05.
<sup>a</sup> F test statistic.

2.3.2. Composition Multivariate compositional analyses were undertaken for both the vegetation and ant datasets. Ant abundances were 4th root transformed but no transformation was applied to vegetation data as they were recorded in a Braun-Blanquet ordinal coverabundance scale (Westhoff & van der Maarel, 1978). Analysis of assemblage composition is recognised as yielding information about ecosystem function that may be overlooked by assessing species richness alone, due to species complementarity and redundancy within an ecological community (Hooper et al., 2005). For the full-width dataset, differences in sampling effort between sites of differing width were accounted for by constructing a similarity matrix through averaging individual pairwise Bray Curtis similarity indices between transects belonging to different sites. No adjustments were made for streamside data because sampling effort was consistent between sites. Preliminary ordinations revealed two outliers in the multivariate space. These sites were identified as having been burnt by wildfires in 2002 and were therefore excluded from multivariate analyses.

Distance-based Linear Modelling (DISTLM) (McArdle & Anderson, 2001) was used to test statistically the combination of variables that best explained the patterns observed in the biotic datasets. A stepwise selection procedure based on the AIC measure of fit was used to identify the optimal sets of environmental explanatory variables within the PRIMER statistical package (PRIMER-E Ltd., Lutton, Ivybridge, UK). Where applicable, DISTLM results were displayed visually via distance-based redundancy analysis (dbRDA) plots (Legendre & Anderson, 1999), with the dominant predictor variable vectors overlaid. These plots are constrained ordinations of the fitted values from multivariate regression models, with axes produced to maximise their linear relationship to fitted values. In the case where a single environmental variable was related to biological data, Principal Coordinates Analysis (PCO) ordination (Gower, 1966) was used to display in two dimensions multivariate differences between sites, with bubbles of different sizes displaying differences in the predictor variable. In addition, the environmental factors that exert most influence on the proportion of exotic to native plant species were also identified using univariate GLM analysis.

## 3. Results

## 3.1. Species richness

The optimal combination of variables to predict average transect species richness of ants (full-width) was (i) site gradient, (ii) soil type, (iii), percent adjacent land as road, (iv) aspect (north–south), (v) perimeter:area ratio and (vi) bush regeneration quality. This model achieved an AIC of 74.26. The removal of the 'Aspect' term however resulted in only a small increase in the AIC value (75.48) yet was preferred because of its greater parsimony (see Table 2). When only streamside transects were considered, a similar suite of variables were found to contribute to ant species richness (see Table 2). The most parsimonious model was(i) gradient, (ii) perimeter:area ratio, (iii) soil type and (iv) corridor width (log), which achieved an AIC of 85.51.

The variables that best predicted the average transect species richness of plants (all species) across the full-width of the riparian corridor were (i) vegetation type, (ii) gradient, (iii) perimeter:area ratio and (iv) corridor width (log). This achieved an AlC of 110.87. The optimal combination of variables for predicting average transect species richness of native plants (full-width) was (i) pH, (ii) area (log), (iii) % adjacent buildings, (iv) % adjacent road, (v) % adjacent pavement and (vi) % adjacent trees/bush, resulting in an AlC

Table 2

A summary of the five best models for predicting the species richness of the streamside and full-width biological datasets. Smaller AIC values result from superior goodnessof-fit and fewer explanatory variables

Ant richness (Transect Avg.)		All plants richness (Transect Avg.)		Native plants richness (Transect Avg.)	
Terms	AIC	Terms	AIC	Terms	AIC
Full width					
Equation (1)	74.26	Equation (2)	110.87ª	Equation (3)	120.52
<ol><li>Aspect (north-south)</li></ol>	75.48ª	(2) + % Adjacent Trees	112.51	(3) – % Adjacent pavement	121.09
(1) + Bush regeneration time	76.24	(2) + Connectivity	112.80	(3) – % Adjacent trees	121.23
(1) – Soil type	78.03	(2) + % Adjacent pavement	112.86	(3) – % Adjacent build – % Adjacent road – % Adjacent pavement – % Adjacent trees	121.24 <sup>a</sup>
(1) – Bush regeneration quality	78.93	(2) - Corr. width + % Adjacent trees	113.22	(3) – % Adjacent pavement + Connectivity	121.88
Streamside					
Equation (4)	84.88	Equation (5)	97.99	Equation (6)	136.71
(4) – Bush regeneration time	85.51ª	(5) + Connectivity	99.96	(6) – pH	136.74 <sup>a</sup>
(4) – Soil type	85.57	(5) + Connectivity – Perimeter: area Ratio	100.65	(6) + % Adjacent water	138.31
(4) + Aspect (east-west)	86.14	(5) - Perimeter: area ratio	100.94 <sup>a</sup>	(6) + Log. corridor width	138.53
(4) + Fire Intensity	86.44	(5) – % Adjacent grass	101.33	(6) + % Adjacent road	138.57

Equation (1): Avg. ant richness ~ site gradient + soil type + % adjacent road + aspect (north–south) + perimeter: area ratio + bush regeneration quality Equation (2): Avg. plant richness (all spp) ~ vegetation community type + site gradient + perimeter: area ratio + corridor width (log).

Equation (3): Avg. plant richness (native spp)  $\sim$  pH + area (log) + % adjacent buildings + % adjacent road + % adjacent pavement + % adjacent trees/bush. Equation (4): Ant species richness  $\sim$  site gradient + perimeter: area ratio + soil type + bush regeneration (time) + corridor width (log).

Equation (5): Plant species richness (all spp)~soil type+% adjacent pavement+gradient+% adjacent grass+perimeter:area ratio+vegetation community type+stream order + % adjacent trees.

Equation (6): Plant species richness (native spp)~ time since fire + pH.

Note: When the two recently burned sites were excluded from Equation (6), only pH was included in the model (AIC 121.6).

<sup>a</sup> Denotes model with the fewest explanatory variables.

of 120.52. Interestingly, removal of all adjacent landscape factors (variables iii-vi) only marginally increased the AIC to 121.24 (see Table 2), indicating that pH and area are the dominant environmental controls on native plant species richness. When only streamside vegetation was considered, some of the variables found to predict full-width plant species richness were selected in the model for streamside plant species richness (all species), with the addition of some other factors. The optimum combination of variables was (i) soil type, (ii) % adjacent pavement, (iii) gradient, (iv) % adjacent grass, (v) perimeter: area ratio, (vi) vegetation community type, (vii) stream order and (viii) % adjacent trees. Given the large number of variables, we chose to omit the perimeter: area ratio term as it contributed least to the model and resulted in only a minor increase in AIC from 97.99 to 100.94 (see Table 2). Like the analysis of full-width native plant richness, modelling the environmental predictors of streamside native plant species richness also identified pH as an important variable, and was retained as the sole predictor when recently burned sites were removed from the analysis (see Table 2).

### 3.2. Composition

### 3.2.1. Ants

The optimal combination of environmental variables found to explain differences in ant communities sampled from the fullwidth of riparian corridors was (i) the percentage of adjacent road, (ii) vegetation community type, and (iii) the connectivity between reserves (Table 3, Fig. 4a). When streamside ant communities were considered in isolation, vegetation community type and the connectivity between reserves were again selected as important environmental predictors, yet the optimum model also included the time since any bush regeneration commenced at a site and the stream order of the adjacent watercourse (Table 3, Fig. 4b).

### 3.2.2. Plants

When all environmental variables were available for inclusion in DISTLM, "corridor width" was the single variable explaining the greatest amount of variation (18.37%) in the vascular plant assemblages (natives and exotics) considered across the full-with of the riparian corridor (Table 3, Fig. 5a). To ascertain whether differences in vegetation composition are driven by the introduction of weeds,

we included as an additional explanatory variable the number of exotic plant species as a percentage of the total plant species richness. This ratio was subsequently an even stronger predictor of assemblage composition than 'corridor width', explaining 23.66% of total variation in plant composition between sites (Pseudo F = 4.34, P=0.001). The optimal suite of environmental variables for predicting the percentage of exotic to native plant species was area (log) + pH (AIC = 150.78).

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When only streamside vegetation was considered, soil pH, not corridor width, was identified as the primary predictor of assemblage composition, and was found to explain 16.66% of the variation (Pseudo F=2.80, P=0.001) (Table 3, Fig. 5b). Again, when the proportion of exotic plant species was included in the streamside model, it was identified as a superior predictor of vegetation composition (Pseudo F = 3.76, P = 0.001).

## 4. Discussion

Our findings both affirm and challenge the current paradigm that corridor width should form the basis of riparian protection policy. Although corridor width was identified as an important variable explaining the biodiversity of riparian corridors, many other environmental factors were found to influence ant and plant communities. Further, the suites of variables differed greatly with respect to the taxonomic group of interest (i.e. ants vs. plants) and the diversity characteristic in question (i.e. species richness vs. composition), highlighting the inherent ecological complexity of riparian corridors

### 4.1. Environmental influences on biotic assemblages

### 4.1.1. Plants

Several mechanisms could explain the observed relationship between corridor width and the composition of vegetation within riparian corridors as a whole. First, wider corridors intrinsically contain a greater proportion of upland habitats, which, in this study area, are characterised more strongly by sandstone-dominated sclerophyllous vegetation than the mesic vegetation found along the streambank. This is highlighted by the fact that corridor width was not selected as a predictor of streamside vegetation composi-

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Details of DISTLM analyses outlining the individual and combined predictive influence of selected environmental variables on ant and plant composition.

	Variable	Individual tests			Sequential tests			
		% Variation explained	PseudoF	Р	Cumulative % variation explained	Cumulative AIC	Pseudo F	Р
Full width								
Ants	% Road	14.64	2.40	0.004	14.64	123.39	2.40	0.001
	Vegetation community type	8.79	1.35	0.119	25.33	123.25	1.86	0.011
	Connectivity	12.21	1.95	0.008	35.07	123.01	1.80	0.022
Plants	Corridor width	18.37	3.15	0.001	18.37	127.31	3.15	0.002
Streamside								
Ants	Vegetation community type	13.42	2.169	0.02	13.42	119.38	2.169	0.022
	Connectivity	11.87	1.8859	0.051	26.14	118.84	2.2402	0.026
	Regeneration time	11.53	1.8253	0.069	37.90	118.06	2.2718	0.023
	Stream order	10.02	1.5594	0.133	45.93	117.85	1.6347	0.11
Plants	pH	16.66	2.80	0.001	16.66	125.66	2.80	0.001



Fig. 4. dbRDA plots of the fitted DISTLM models for the ant dataset, with overlays of environmental variables. Panel (a) shows the response for full-width data, and panel (b) displays streamside data only. "Veg. Community Type" is the type of mapped vegetation community the site is situated within, "% Road" is the proportion of adjacent land occupied by paved roads, "Connectivity" is the degree to which the site is linked with other larger reserves, "Stream Order" indicates stream size based upon a hierarchy of tributaries, and "Regen. Time" is the time since any bush regeneration activities commenced on a site.

tion. Second, edge effects may be more influential in narrow than wide corridors. The degree of edge influence is related broadly to the proportion of edge to interior habitat (Forman, 1995). In the case of riparian corridors, edge effects are derived from both the stream (flow-derived disturbance) and the urban boundary (e.g. enhanced levels of nutrients, pollutants, light, and physical distur-



Fig. 5. PCO plots of the response in vegetation composition to the primary environmental predictor. Panel (a) is the response of vegetation located throughout the entire riparian corridor to the width of that corridor. Panel (b) displays the response in streamside vegetation to soil pH. Each bubble represents a site, and its size is proportional to its riparian corridor width. Bubbles closer together represent sites more similar in composition.

bance). It is likely that many urban riparian corridors do not include any true 'interior' habitat because of the significant environmental stresses placed upon them. Nevertheless, their geometry is likely to influence the magnitude of these impacts on biotic communities. Indeed, the fact that riparian corridors can be irregularly shaped (Soule & Gilpin, 1991) may have emphasised the edge influence of the adjacent landscape on the vegetated riparian corridor (Forman, 1995), thus resulting in the identification of "perimeter: area ratio" as a significant predictor of full-width plant species richness at our sites (Table 2).

When the effect of lateral environmental variation on vegetation assemblages was removed by studying streamside vegetation in isolation, soil pH was identified as being an important predictor of riparian vegetation composition. This corresponds strongly with models of native plant species richness that also included soil pH as a predictor variable (Table 2). Soil pH is a likely predictor of riparian vegetation because of its influence on nutrient availability and the toxicity of metal ions in the soil (Townsend, Begon, & Harper, 2003, p. 116). Accordingly, pH may be either positively or negatively related to plant species richness (Dupré, Wessberg, & Diekmann, 2002) depending on the soil pH and vegetation type. Urbanisation has been associated with increased alkalinity of riparian soils (Moffatt et al., 2004), thus soil pH may be a proxy for urban development intensity. One potential reason for this is the dissolution of cement products into stormwater conveyed by concrete gutters and pipes (Davies, Wright, Jonasson, & Findlay, 2010). In addition, soil pH is associated intrinsically with the underlying geology. Hawkesbury sandstone soils are generally nutrient-poor, acidic and floristically species rich in comparison to the nutrient-rich and alkaline Wianamatta Shale derived soils (Chapman & Murphy, 1989). The negative relationships between native plant richness and soil pH for both streamside and full-width vegetation (Table 1) are likely to be a reflection of this geological pattern, but may also be compounded by the fact that the most heavily urbanised areas in Ku-ring-gai are found on alkaline shale ridgetops that also experience greater weed invasion pressure (Lake & Leishman, 2004).

Studies that have looked explicitly at patterns of weed invasions have found clear links between soil pH and the species richness of alien plants (e.g. Alston & Richardson, 2006). Furthermore, research from the Sydney region has shown that alien species richness is negatively associated with the number of native species (King & Buckney, 2002). Disturbed and nutrient enriched environments are often particularly susceptible to invasion because the competitive advantage afforded to many alien species by these factors (Davis, Grime, & Thompson, 2000; King & Buckney, 2002) can lead to the exclusion of native species and a reduction in overall community species richness (e.g. Gerber, Krebs, Murrell, Moretti, & Schaffner, 2008; Moffatt et al., 2004). It can be difficult to determine without manipulative experimentation if the presence of alien species per se results in the loss of native species, or whether alien species take advantage of environmental conditions that are unfavourable for native species (Bridgewater & Backshall, 1981). Nevertheless, the studies listed above indicate that the variability in the richness and composition of vegetation assemblages observed in the present study may be driven predominantly by the presence and dominance of exotic species. Indeed, as streamside habitats are notoriously dominated by weeds in the Sydney region (Lake & Leishman, 2004), the area and corridor width terms identified in the full-width models of vegetation richness and composition respectively suggests that these merely function as a proxy for the proportion of streamside to hillslope vegetation.

### 4.1.2. Ants

Ant assemblages appear not to be influenced as strongly by streamside-hillslope environmental gradients as vegetation assemblages, given the inclusion of many of the same environmental predictors in streamside and full-width analyses (see Tables 2 and 3). However, ants were influenced by a more complex suite of environmental variables, namely road density, vegetation community type and reserve connectivity. The density of road networks is a good proxy for urbanisation intensity (Brady, McAlpine, Miller, Possingham, & Baxter, 2009), which has been related to butterfly species richness (peaking at moderate urbanisation levels) (Blair & Launer, 1997) and a simplification of ant communities (Thompson & McLachlan, 2007). As higher urbanisation intensity is associated with smaller backyards and fewer urban green spaces, there may be reduced invertebrate dispersion from the urban matrix to riparian corridors in these areas (cf. Sperling & Lortie, 2010). The fact that road density was identified in the assessment of full-width but not streamside data suggests that the influence of the adjacent landscape on riparian corridors may be greatest nearest the urban boundary, and that streamside habitats are buffered from this influence in wider corridors. Our findings therefore align with current empirical evidence of the influence of landscape urbanisation on invertebrate communities.

The significant influence of vegetation community type on ant assemblages is consistent with the findings of a number of previous studies. Ant communities are known to respond to vegetation differences (Andersen & Majer, 2004), yet can exhibit higher rates of turnover than plant communities (Oliver, Beattie, & York, 1998). Ants may respond to both the environmental (soil, geology, slope, etc.) and floristic characteristics that dictate vegetation community types, as these are also known to influence nest establishment, food supply and interspecific dynamics (Carroll & Janzen, 1973; Ryti & Case, 1992). Our research confirms that ants are sensitive to floristic and environmental variation as they were correlated with small differences in vegetation type (all vegetation was broadly eucalypt forest).

The principles of island biogeography (MacArthur & Wilson, 1967) may explain the influence of reserve connectivity on ant assemblages. A tract of streamside bushland may be considered as an isolated habitat island within an urban landscape. The composition of ant communities within will likely be a function of the size of the reserve and its distance from source populations according to available niche space and species' range requirements (cf. Gibb & Hochuli, 2002). The distance of a patch from larger reserves influences both the immigration rate of individuals as well as the type of species immigrating based upon their dispersal traits (see Pulliam, 1988). Therefore, ants in disconnected corridors are likely to contain either (i) populations that can survive in small patches that have a high degree of edge effects, or (ii) populations that are maintained constantly by dispersal from larger areas, with a bias towards species with good dispersal ability. Ants with these life history characteristics generally fall into the category of 'opportunists' (Andersen, 1995) and have been found to dominate ant fauna within urban areas (Gibb & Hochuli, 2002; Lessard & Buddle, 2005). Hill (1995) provided empirical evidence that vegetated corridors are dispersal conduits for invertebrates when he observed interior-habitat specialist-species of ants, dung beetles and butterflies in linear strips of riparian vegetation that linked larger reserves, but not in adjacent arable land.

### 4.2. Considerations

Although a range of environmental factors that influence the biodiversity of streamside corridors were identified, it is likely that synergistic interactions exist between the measured variables and others that were not measured in this study. For example, corridor width may influence vegetation composition more significantly within catchments that are more densely developed. Furthermore, factors such as microhabitat characteristics are also likely to influence ant and plant biodiversity characteristics. For example, ant 380

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species richness and composition vary with soil moisture and texture (Johnson, 1992; Levings, 1983) and the extent of surface litter and coarse woody debris (Lassau & Hochuli, 2004; Thompson & McLachlan, 2007). It would be worthwhile considering such variables in any future studies. It is also worth noting that since our entire study was located within the greater metropolitan area of Sydney, some degree of ecological change resulting from urbanisation is likely to have already influenced all biotic communities we sampled (cf. McKinney, 2002), yet the scale at which our study was undertaken did not permit these broad ecological changes to be analysed.

### 4.3. Implications for riparian conservation and management

The identification of corridor width as the best environmental metric to predict vegetation composition across the full width of riparian corridors is tempered by the fact that this relationship is driven predominantly by the presence of novel habitats upslope in wider corridors. Although protecting these upslope communities is not without merit from a landscape diversity perspective, it appears that wider corridors do not necessarily improve the condition of streamside vegetation per se. Instead, the influence of site-specific environmental factors such as soil pH on streamside vegetation composition implies that environmental planners should strategically protect riparian bushland located in different environmental settings in order to enhance vegetation diversity at the catchment scale.

Furthermore, since variation in riparian vegetation is largely a function of the presence and dominance of exotic species, management strategies that focus specifically on weed control may also be of value. Identification of greater weed dominance in alkaline soils can also assist managers in targeting sites at risk of invasion. Although weeds can provide some functional value within an ecosystem (e.g. Shapiro, 2002), their homogenising impacts on species diversity and degradation of ecosystem function (Levine et al., 2003) should make them a primary issue in riparian management. Applying restoration techniques and maintaining native vegetation complexity along reserve edges (cf. Cadenasso & Pickett, 2001) is likely to enhance biodiversity outcomes across a full range of corridor widths. Nevertheless, we recommend that riparian corridors be designed to minimise the proportion of edge to interior habitat irrespective of width, in order to minimise negative edge effects. This should not, however, work against the primary aim of retaining as much bushland within suburbia as possible. Rather, where trade-offs are necessary, priority should be placed on configuring reserves so as to minimise convoluted boundaries and associated edge effects.

The response of ant communities to a different number and set of environmental variables than plants suggests that management approaches formulated for one taxonomic group will not necessarily translate into appropriate outcomes for another. This should be remembered when selecting habitat patches to function as target communities for biodiversity strategies. Furthermore, a focus on maximising corridor width may not guarantee effective protection of ant communities. Although many studies have investigated the optimum dimensions for conservation of a range of taxa such as amphibians and reptiles (Semlitsch & Bodie, 2003), birds, mammals and plants (Spackman & Hughes, 1995), data on invertebrates are limited. Our findings indicate that policies and strategies that take into account variability in catchment characteristics and matrix habitat (e.g. the degree of urbanisation), the location of other reserves of conservation significance, and the type of vegetation community present are necessary for invertebrate conservation.

In practice, this landscape-scale riparian management approach may involve emphasising the retention of bushland that is well connected and close to large reserves, or prioritising that which

contains a vegetation community poorly represented in the existing reserve network. Additionally, the impact of the urban matrix on remnant bushland can be minimised through careful urban planning and ecologically sensitive management of private and public land, and has the potential to 'widen' riparian corridors in an ecological sense (Parker et al., 2008). Application of Water Sensitive Urban Design principles, the integration of greenspace and planting of indigenous vegetation on private lands (Parker et al., 2008) are all examples of management approaches that can collectively alleviate detrimental urban pressures at both the site and catchment scales thereby enhancing biodiversity outcomes across the full spectrum of riparian corridor widths.

## 5. Conclusions

This study has demonstrated that in comparison with other environmental factors, corridor width does not have a large effect on the biodiversity of riparian corridors, particularly when the natural variation in habitat from the stream bank upslope is discounted. Other environmental variables (e.g. soil pH, roads, reserve shape and connectivity) that are commonly given less consideration in riparian policies appear also to have a significant effect on biotic assemblages, yet differ in their influence between ant and plant communities. Riparian management policies should focus therefore on actively improving the condition of riparian bushland (e.g. managing the dominance of invasive species) and reducing the impact of surrounding land uses (e.g. maximising connectivity between reserves and minimising the intensity of development at the watershed scale). However, effective outcomes will only be achieved by the consideration of scientific knowledge within early stages of riparian policy formulation, and the subsequent integration of these policies into broader environmental planning instruments.

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# CHAPTER 5

# The Influence of Catchment Land Use on Riparian Ant Diversity

**Paper:** "Comparison of riparian ant assemblages between rural and urban catchments"

CD Ives, MP Taylor, DA Nipperess and GC Hose In preparation for the journal *Landscape Ecology* 

The incomplete account of between-site variance in ant assemblages by environmental variables in Chapter 4 along with the observed influence of adjacent roads prompted a more targeted assessment of the effect of catchment land use on riparian biodiversity. As road density is widely considered a proxy for urbanisation intensity, the impact of broadscale development on riparian ant assemblages was investigated via comparison of selected sites in urban and rural settings. At all sites, environmental and vegetation data were collected to help identify the mechanisms that may contribute to patterns in ant assemblages. Given the significance of riparian corridor width identified in Chapter 3, the configuration of riparian reserves were controlled for as much as possible between the two land uses. Results from this study are used to provide recommendations for the management of riparian zones in the context of urban development.

# Author Contributions:

# **CD lves: 85%**

Conception of hypothesis; study design; formulation of methodology; fieldwork and data collection; statistical analysis and interpretation; writing and editing of text.

## MP Taylor: 5%

Advice with study design; editing of text.

# DA Nipperess: 5%

Assistance with statistical analysis; editing of text

# GC Hose: 5%

Advice regarding study design; assistance with statistical analysis; editing of text.

# Abstract

The conversion of rural land to residential use is occurring rapidly as the global trend of city expansion continues. The effects of this land use change on riparian biodiversity was investigated by examining ant communities in 24 remnant riparian corridors in northwest Sydney, Australia. Terrestrial macro-invertebrates were sampled from rural and urban catchments using pitfall traps, as well as data on vegetation floristics, habitat structure and stream water quality. Contrary to predictions, no significant differences in ant diversity metrics or assemblage structure were found between urban and rural sites. Similarly, very few environmental variables revealed significant differences between land use types. One exception, however, was the cover of alien plants, which was also related to ant assemblage structure. Small-leaved and broad-leaved privet (Ligustrum spp.) were highly correlated with total weed cover, which was significantly related to ant composition. The results suggest that ants respond mostly to small-scale habitat characteristics, especially those related to the dominance of exotic plants. As no unperturbed catchments were sampled, it is possible that the sites may have transgressed an ecological threshold following earlier land use change to agricultural use, thus explaining the failure to observe differences between urban and rural catchments. We recommend that managers seeking to protect riparian invertebrate communities in urban and rural landscapes should concentrate on weed abatement and consider both biotic and abiotic causes of invasion.

# Key Words

Urbanisation, catchment land use, riparian corridors, biodiversity, ants, weed invasion, spatial scale.

# 1. Introduction

Urbanisation is one of the most significant processes currently transforming landscapes on a global scale. As human populations become increasingly urbanised (United Nations 2008), urban sprawl is leading to the replacement of greenfield sites with low/medium density housing. Indeed, this 'exurban' development (6 – 25 homes per km<sup>2</sup>) is the fastest growing land use in the USA (Brown et al. 2005). Increases in the proportion of sealed surfaces diminish infiltration and evapotranspiration and lead to greater conveyance of runoff to natural waterways. This results in a reduction in baseflow and more frequent, short duration, high volume discharges (Burton et al. 2005; Paul and Meyer 2001). These hydrological changes have been found to impair aquatic ecosystems (Allan 2004; Walsh et al. 2005b) and riparian ecological processes (Groffman et al. 2003). Scarce research has been conducted on the impacts of urbanisation on riparian invertebrate communities. One recent study in northern Sydney Australia, however, found that communities of ants in riparian corridors were influenced by adjacent land use (Ives et al. 2011a).

Across many human-altered landscapes, riparian corridors are important for the conservation of biodiversity (lves et al. 2011b; Lees and Peres 2008; Naiman et al. 1993), the maintenance of river geomorphic integrity (Simon and Collison 2002) and buffering waterways from non-point source pollutants and eroded soil material (Dosskey et al. 2010). Recent advances in the protection offered to riparian corridors in legal statutes (Davies et al. 2011; Lee et al. 2004; Wenger and Fowler 2000) has established them as important environmental assets in both urban and rural

landscapes. In addition, riparian corridors are ideal systems in which to study the ecological effects of urbanisation because they can reflect changes catchment and stream condition because of their position between aquatic and terrestrial environments.

The majority of studies investigating arthropod responses to urbanisation have described significant but variable responses in biodiversity or assemblage structure with anthropogenic disturbance (Hochuli et al. 2009 and references therein). For example, Sadler et al. (2006) identified greater species richness of carabid beetles in rural areas than in urban landscapes, while Sanford et al. (2009) found species richness and abundance of ants to peak at intermediate levels of development. Interestingly, Lessard and Buddle (2005) found a higher richness of ants species in urban backyards than in remnant forest interiors, albeit with the addition of more aggressive and dominant species at urban sites. However, no studies have examined specifically responses in riparian invertebrate assemblages as a result of urban land use transformation.

In this study, ants were selected to assess ecological changes associated with urbanisation because they are ubiquitous in the Australian landscape (Andersen and Majer 2004), have high levels of species diversity (Andersen 1995b), are easily sampled (Andersen et al. 2004) and respond rapidly to changes in environmental conditions (Andersen et al. 2004; McIntyre 2000; Pik et al. 2002). Ants have been found to respond to both landscape scale pressures such as reserve size (Gibb and Hochuli 2002; Yamaguchi 2004) and small-scale microhabitat features such as soil

moisture, leaf litter and tree cover (Lassau and Hochuli 2004; Vepsalainen et al. 2008). They also play an important role in the function of ecosystems through their role in nutrient cycling (Folgarait 1998), soil development (Richards 2009), aiding runoff infiltration (Sanford et al. 2009), seed dispersal (Rico-Gray and Oliveira 2007), and via predation on other insects as well as being food for other species (Folgarait 1998; Hölldobler and Wilson 1990). Consequently, ants have been used widely as indicators of ecological condition and ecosystem restoration and function (Hoffmann and Andersen 2003; Lomov et al. 2009; Sanford et al. 2009).

Using sites from north-west Sydney, Australia, we sought to evaluate differences in riparian ant assemblages due to the transformation of low-density rural/residential land to higher density residential housing. This study conceptually follows lves et al. (2011a), where relationships were identified between the diversity of riparian ant communities and characteristics of the surrounding landscape within the Ku-ring-gai LGA, Sydney. North-west Sydney provided an opportunity to assess these previously identified trends more rigorously, through studying the ant diversity of remnant riparian corridors situated in two proximal but contrasting land-use types. We predict that riparian ant assemblages in urban catchments have reduced species diversity compared with rural catchments, and that the two groups are significantly different in composition. Two processes were postulated to give rise to these patterns. First, since the hydrology and water quality of a stream carries the 'signature' of the catchment land-use, we expected that higher discharges, flow rates and contaminants associated with urban streams would place additional stress on riparian ant communities. Secondly, we anticipated that riparian zones in urban

catchments would experience greater direct impacts from the adjacent landscape than rural sites because of greater human interaction, diffuse pollutants and other edge effects related to residential land use.

# 2. Methods

## 2.1. Survey Design

Sites were located in the north-west of Sydney within The Hills, Hornsby and Ku-ringgai Local Government Areas (Latitude = 33°37'32"S - 33°44'40"S; Longitude = 151°08'39"E - 150°57'55"E) (Fig. 1). Agricultural and pasture land has been urbanised progressively over the past 25 years within restricted land use zones in these areas. To test for differences in ant assemblages in response to land use, six riparian sites were selected from within rural-dominated catchments and six from urban-dominated catchments. Rural and urban catchments differed markedly in a number of quantitative measures of urbanisation such as road density (Rural = 18–55 m/ha; Urban = 153-186 m/ha), property density (Rural = 0.72-1.43 lots/ha; Urban = 9.25-12.75 lots/ha) and average lot size (Rural = 13,335-17,617 m<sup>2</sup>; Urban = 1133-1484 m<sup>2</sup>). We were careful to match as many landscape variables as possible across the two land uses because anthropogenic and natural landscape features often covary (Allan 2004). Sites were located on public and private land within contiguous areas of riparian bushland of consistent width. They were adjacent to low to mid order streams (Strahler 1952) with upstream catchment areas of between 72 ha and 363 ha. All sites were on Hawkesbury Sandstone and were part of the Sydney Sandstone Gully Forest vegetation unit (Benson and Howell 1994). Land use of the

surrounding subcatchment was relatively homogeneous, with the majority of land in the rural catchments zoned as "Rural" land and the urban catchments as "Residential" land under respective Local Environment Plans (e.g. The Hills Shire Council 2011). Areas of active bushland restoration were avoided, as were any other visible signs of physical disturbance.

At each site, terrestrial macro-invertebrates were collected along with data on vegetation floristics and structure, physical habitat characteristics, water chemistry and geomorphology of the adjacent stream (Table 1).



**Figure 1** – Location of study sites within the north-west of Sydney. Sites beginning with "R" are within rural catchments. Sites beginning with "U" are within urban catchments.

VARIABLE	UNITS	CODE	RANGE	TRANS- FORMATION	-	ס
Number of Strata	Integer	Num_Strata	3–4		0.542	0.599
Foliage Cover of Upper Stratum	%	FC_Upper1	50-70		0.222	0.828
Foliage Cover of Mid Stratum	%	FC_Mid	40–70		2.58	0.027*
Foliage Cover Lower Stratum	%	FC_Lower	20-85		1.051	0.318
Growth Forms	PCA Axis 1 Values	GF_All	-52.46- 22.72		1.51	0.163
Height Upper	ш	HeightUpper	15-35		0.576	0.577
Height Mid	Э	HeightMid	2-8		0.157	0.878
Height Lower	Э	HeightLower	0.1-0.5		0.786	0.45
Proportion of Weed Cover (all strata)	%	WeedCovAll	3–70	log	2.939	0.015*
Proportion of Weed Cover (mid stratum)	%	WeedCovMid	0-80	Square root	1.895	0.087
Proportion of Weed Cover (lower stratum)	%	WeedCovLower	5-50	Square root	1.761	0.109
Weed Species Richness (S) as a proportion of Total Plant S (all strata)	%	WeedRichAll	3-30	log	3.237	0.009*
Proportion of Weed S to Total Plant S (mid stratum)	%	WeedRichMid	0-50	Square root	1.859	0.093
Proportion of Weed S to Total Plant S (lower stratum)	%	WeedRichLower	0-30	Square root	0.557	0.59
Leaf Litter	%	LeafLitter	15–50		1.12	0.29
Rocks Logs and Debris	%	RLD	5–15		1.0	0.341
Soil Texture (Sand/Silt/Loam/Clay)	PCA Axis 1 Values	SoilText	-8.33 91.67		1.0	0.341
Soil pH	pH Units	Soil_pH	4.7–6		1.23	0.246
Water pH	pH Units	Water_pH	6.4-7.25		0.586	0.57
Water temp	Degrees C	Water_Temp	17.3–21.7		0.596	0.564
Water Electrical Conductivity	MicroSiemens	Cond	298–979	log	0.872	0.404
Water Salinity	ppm	Salin	165–504	log	0.975	0.352
Water Dissolved Oxygen	%	DO	0-92.6		0.579	0.575

study sites are included in columns 't' and 'P'. Table 1 Table of environmental variables. Results of t-tests between environmental variables from urban and rural

\* Significant at  $\alpha = 0.05$
## 2.2. Field Sampling and Invertebrate Processing

Field sampling was undertaken between the 17<sup>th</sup> and 24<sup>th</sup> of March 2010. Ten 250 mL pitfall traps (diameter 70 mm) were installed at each site. These were arranged along a 50 m linear transect positioned ~5 m from the top of bank, parallel to the stream. Traps were filled with a mixture of 70:30 ethanol:water to kill and preserve specimens as well as a small amount of detergent to break surface tension. Pitfall traps have been found to capture a good representation of invertebrate diversity compared with other sampling techniques, and as such are used widely in field sampling of ants (Andersen 1991). Traps were left open for 7 days.

The structure of the vegetation 5 m either side of the transect was recorded at each site, and for each stratum the following measurements were taken: (i) average height, (ii) growth form (i.e. tree, shrub, herb etc.), (iii) total foliage cover, (iii) dominant vascular plant species and their cover, (iv) the proportional cover of weeds and (v) the proportion of weed species to the total number of species at a site. A seven-class Braun-Blanquet cover abundance scale was used to quantify rapidly species cover. Weed proportions were derived from visual estimates to the nearest 5% (Hill 2005). All records were collected by the lead author to maintain consistency and reliability of data.

Techniques for measuring local habitat characteristics were adapted from Lassau & Hochuli (2004) and involved assigning a score for the per cent cover of rocks/logs/debris and leaf litter (4 classes: 0; <30; 30-70; >70%) and soil moisture (3 classes: dry; moist; wet). Soil field texture was also recorded along with soil pH, as

indicated by a field test kit. Water temperature, pH, conductivity, salinity and dissolved oxygen (DO) of the stream adjacent to invertebrate survey transect were measured using hand held probes (DO/Temp: YEO-KAL 609; conductivity/salinity: Hanna HI9835; pH: Hanna HI9125).

Ant specimens were identified to Genus using the key in Shattuck (1999) and were split subsequently into morphospecies, hereafter referred to as 'species' for simplicity. In addition, ant species were grouped into functional groups according to their role in the ecosystem (Andersen 1995a).

#### 2.3. Statistical Analyses

#### 2.3.1. Data Pre-treatment

At a number of the sites, pitfall traps had been tampered with or were missing, most likely due to animal disturbance. The minimum number of complete pitfall samples at a site was 7 (site U4). The resulting unequal sampling effort between sites was accounted for via performing a rarefaction procedure performed on site data in the R statistical environment (R Development Core Team 2011). This procedure involved selecting 7 samples at random from each site over 1000 permutations, and averaging the species abundances to obtain a representative species abundance dataset for each site given a sampling effort of 7 pitfall traps. The large number of environmental variables collected were reduced in number through performing a PCA and recording the primary axis as a single representative variable. For example, taking the Axis 1 of a PCA on water temperature, pH, conductivity, salinity and dissolved oxygen enabled a single variable 'water quality' to be derived. These summarised variables were used in analyses of relationships between environmental data and species-abundance datasets.

A number of variables were not normally distributed and were thus transformed prior to statistical analysis. Log<sub>10</sub> transformation was applied to the proportional cover of weeds (all strata), proportional weed species richness (all strata), and the conductivity and salinity of the stream. Square root transformation was applied to the proportional cover of weeds (mid and lower stories), proportional weed species richness (mid and lower stories) and the per cent dissolved oxygen in the stream. All environmental variables were normalised prior to multivariate analysis to correct for differences in measurement units.

#### 2.3.2. Statistical Techniques

#### a) Comparisons Between Rural and Urban Sites

Site species richness, abundance, Shannon Diversity index and Pielou's Evenness index were calculated from the rarefied ant dataset (see Magurran 2004 for formulae). Differences in these diversity metrics between rural and urban land uses were assessed using t-tests, as were differences in vegetation and environmental data. Ant and plant compositional differences between land use types were assessed using PERMANOVA (Anderson 2001) in the PRIMER statistical package (PRIMER-E

Ltd, Lutton, Ivybridge, UK). This technique uses a permutation procedure to test for differences between samples based upon any resemblance measure. Like standard ANOVA, it can also account for the influence of co-variables on a response dataset. Compositional differences in ant communities between sites were displayed visually using Principal Coordinates Analysis (PCO) ordination (Gower 1966).

#### b) Environmental and Floristic Influences on Ant Communities

Pearson correlation coefficients were calculated to test for statistically significant relationships between ant diversity and environmental variables. Influences of different environmental variables on ant species composition were assessed using Distance-Based Linear Modelling in PRIMER: a multivariate procedure that tests for a relationship between linear predictive variables and community response data through partitioning of variation according to a regression model (McArdle and Anderson 2001). Environmental variables found to be statistically related to ant composition were also superimposed onto PCO plots of ant community data using different sized circles to denote magnitude.

#### c) Community Similarity Between Ants and Plants

Relationships between ant assemblage composition and vegetation composition were assessed using the RELATE procedure, also in PRIMER. This routine calculates a Mantel coefficient (Mantel 1967) based upon the correlation between the pairwise resemblance comparisons between samples for each dataset.

## d) Interactions Between Plants and Ants

Plant species that were most highly correlated with ant community data were identified through DISTLM analysis using the BEST variable selection procedure. In

addition, ant species that were most closely related to weed cover were also identified through Spearman correlation tests between individual ant species abundances and the weed cover at a site.

# 3. Results

A total of 5249 ant species were collected from 119 pitfall traps. Sixty species were identified comprising 34 genera (see chapter appendix).

# 3.1. Comparison of Rural and Urban Ant Communities

There were no significant differences in richness, abundance, Pielou's Evenness or Shannon's Diversity of ant assemblages between urban and rural land uses (Table 2). Similarly, there was no significant difference in ant assemblage composition between catchment land use types (PERMANOVA: Pseudo-F = 1.5863, P = 0.071). The similarity among sites is endorsed by the lack of clear catchment related patterns in the ordination (Fig. 2a). There were also no significant differences in the functional group composition of ants between rural and urban sites (Pseudo-F = 0.236, P =0.916) (see Fig. 2b).

**Table 2** Results of t-tests of univariate diversity measures of ants between urban and rural study sites.

	RU	IRAL	UR	RBAN	T-T	EST
	Mean	St. Dev.	Mean	St. Dev.	t	Ρ
Species Richness	18.5	7.3	15.8	2.7	0.835	0.424
Abundance	358.5	179.6	261.3	140.1	1.045	0.320
Pielou's Evenness	0.5	0.1	0.5	0.2	0.532	0.607
Shannon Diversity	1.5	0.4	1.3	0.5	0.567	0.583



**Figure 2**: Ordinations of ant assemblage data with symbols distinguishing between land use. Figure (a) is for species data, while figure (b) is for ant functional groups. Grey triangles represent rural study sites and black squares represent urban sites.

#### 3.2. Comparison of Rural and Urban Environmental Variables

Few significant differences in local site environmental variables were observed between rural and urban catchments (see Table 1). The exceptions were mid storey vegetation cover (rural mean = 53.3%, st. dev. = 11.25; urban mean = 65.8%, st. dev. = 3.76), the proportional cover of weeds (rural mean = 8.66, st. dev. = 0.31; urban mean = 0.45, st. dev. = 1.41) and the proportion of weed species to site richness (square-root transformed) (rural mean = 0.675, st. dev. = 0.193; urban mean = 1.159, st. dev. = 0.311). Additionally, urban and rural catchments showed no significant differences in the composition of either (i) the dominant vegetation species present at a site or (ii) weed species (dominant vegetation: Pseudo-F = 1.172, P = 0.337; weed species: Pseudo-F = 1.543, P = 0.19).

## 3.3. Other Environmental Influences on Ant Communities

Of all variables tested, only height of the lower stratum and proportion of weeds in the lower stratum were correlated with ant diversity metrics (Table 3). A significant linear relationship was however identified between ant community composition and the cover of weeds at a site (log transformed) (Pseudo-F = 1.93, P = 0.01; Fig. 3). No other variables were significantly related to ant community composition (see Table 4)



**Figure 3** Ordination of ant assemblages overlaid with the proportion of weed cover (bubbles) and vectors of ant species most highly correlated with the proportion of weed species at a site.

\* Significant at  $\alpha = 0.05$ 

					EN	VIF	RON	IME	ENT	AL	AN	D F	LO	RIS	TIC		RI	٩BL	ES								
Water_%DO (Sqrt)	Water_Salinity (Log)	Water_Cond (Log)	Water_temp	Water_pH	Soil_Texture	Soil_pH	Soil_Moisture	%LeafLitter	%RocksLogsDebris	WeedRich%_Lower (Sqrt)	WeedRich%_Mid1 (Sqrt)	WeedRich%_ALL (Log)	WeedCA%_Lower (Sqrt)	WeedCA%_Mid1 (Sqrt)	WeedCA%_ALL (Log)	Height_Lower	Height_Mid1	Height_Upper1	FC%_Lower	FC%_Mid1	FC%_Upper1	GF_Lower1	GF_Mid1	Number_Strata			
0.192	-0.321	-0.310	0.236	-0.044	0.163	0.221	-0.456	0.255	-0.163	0.111	0.120	0.038	-0.139	-0.123	-0.211	-0.596	-0.319	0.088	-0.213	0.138	-0.495	-0.350	0.043	0.264	r	SPE	
0.619	-1.070	-1.030	0.769	-0.138	0.524	0.716	-1.622	0.835	-0.524	0.354	0.381	0.122	0.443	-0.394	-0.683	<u>-2.350</u>	-1.063	0.281	-0.688	0.442	-1.800	-1.181	0.135	0.866	+	CIES RICH	
0.550	0.310	0.327	0 <u>.</u> 460	0 <u>.</u> 893	0.612	0 <u>.</u> 490	0.136	0.423	0.612	0.731	0.711	0.906	0.667	0.702	0.510	0.041*	0.313	0.785	0.507	0.668	0.102	0.265	0.896	0.407	P	INESS	
-0.504	-0.075	0.013	0.037	-0.149	-0.058	-0.428	-0.002	0.427	-0.393	0.200	-0.001	-0.131	0.229	0.240	-0.032	-0.035	-0.396	-0.182	-0.150	0.263	0.019	0.046	0.043	0.425	-	A	
-1.843	-0.238	0.041	0.116	-0.475	-0.184	-1.497	-0.005	1.492	-1.353	0.646	-0.003	-0.417	0.744	0.782	-0.102	-0.111	-1.364	-0.584	-0.479	0.862	0.060	0.145	0.136	1.485	-+	BUNDANC	
0.095	0.817	0.968	0.910	0.645	0.857	0.165	0.996	0.167	0.206	0.533	0.998	0.686	0.474	0.453	0.921	0.914	0.202	0.572	0.642	0.409	0.953	0.888	0.894	0.168	ק	Ť	ANT DIVER
0.258	0.392	0.346	0.451	0.123	0.076	0.218	-0.215	-0.347	0.191	-0.604	-0.551	-0.211	-0.469	-0.515	-0.234	-0.072	0.175	0.223	0.419	-0.420	-0.154	0.233	0.319	-0.125	r	PIELC	SITY INDIC
0.845	1.346	1.165	1.600	0.390	0.242	0.706	-0.696	-1.168	0.616	-2.398	-2.087	-0.683	-1.681	-1.898	-0.763	-0.229	0.561	0.722	1.461	-1.462	-0.492	0.756	1.065	-0.398	+	DU'S EVEN	ËS
0.418	0.208	0.271	0.141	0.704	0.814	0.496	0.503	0.270	0.552	0.037*	0.063	0.510	0.124	0.087	0.463	0.824	0.587	0.487	0.175	0.174	0.633	0.467	0.312	0.699	P	NESS	
0.240	0.174	0.144	0.431	0.013	0.136	0.356	-0.325	-0.199	0.120	-0.476	-0.331	-0.109	-0.252	-0.409	-0.195	-0.240	0.038	0.163	0.293	-0.283	-0.302	0.085	0.310	-0.005	٦	SHAN	
0.782	0.558	0.461	1.508	0.043	0.435	1.205	-1.087	-0.641	0.382	-1.713	-1.109	-0.347	-0.825	-1.419	-0.627	-0.782	0.120	0.523	0.969	-0.935	-1.001	0.269	1.032	-0.017	+	JON'S DIVE	
0.453	0.589	0.654	0.162	0.967	0.673	0.256	0.303	0.536	0.711	0.118	0.293	0.736	0.428	0.186	0.545	0.452	0.907	0.612	0.356	0.372	0.341	0.793	0.327	0.987	ק	RSITY	

 Table 3 Correlations between Ant Diversity and Environmental Variables

Variable	Pseudo-F	Р	% Variance Explained
Number of Strata	0.658	0.856	6.17
Growth Forms	0.937	0.561	8.57
Foliage Cover	0.516	0.966	4.91
Height	1.037	0.421	9.40
WeedCA%_ALL (Log)	1.930	0.010	16.18
WeedCA%_Strata_TRF	1.236	0.210	11.00
WeedRich%_ALL (Sqrt)	1.454	0.094	12.70
WeedRich%_Strata_TRF	0.641	0.887	6.02
Surface Material	0.988	0.457	8.99
Soil_Characteristics	1.451	0.083	12.67
Water_Chemistry_TRF	1.262	0.182	11.20

 Table 4 DISTLM results for Environmental Variables v Ant Community Composition

### 3.4. Community Similarity Between Ants and Plants

Comparison of the ant species dataset with species abundance data of dominant vegetation did not indicate any significant relationship between these two groups ( $\rho = 0.105$ , P = 0.211; Spearman rank correlation). However, a significant relationship between ant and weed assemblages was present ( $\rho = 0.284$ , P = 0.013, Spearman rank correlation).

## 3.5. Interactions Between Weeds and Ants

Because of the significant relationship between ant composition and the cover of weeds, we tested again for differences between urban and rural catchments while accounting for percentage weed cover as a covariate in the PERMANOVA design. However, no observable difference between rural and urban catchments was revealed after controlling for differences in weed cover (weeds: Pseudo-F = 1.866, P = 0.014; land use: Pseudo-F = 1.098, P = 0.360).

*Ligustrum lucidum* (broad-leaved privet) was the exotic plant species most highly correlated with ant composition (Pseudo-F = 1.651, P = 0.038). *Ligustrum sinense* (small-leaved privet) was also highly correlated (Pseudo-F = 1.551, P = 0.07), but other weeds contributed little to explaining patterns in ant composition (Table 5). *Ligustrum lucidum* and *L. sinense* were both related closely to the total percentage weed cover at a site (*L. lucidum*: Spearman  $\rho = 0.74$ , P = 0.014; *L. sinense*: Spearman  $\rho = 0.84$ , P = 0.005).

Variable	Pseudo-F	Ρ	% Variance Explained
Ageratina adenophora	0.723	0.811	6.74
Ageratina riparia	0.895	0.634	8.21
Lantana camara	1.247	0.242	11.09
Ligustrum lucidum	1.651	0.038*	14.17
Ligustrum sinense	1.551	0.07	13.43
Lonicera japonica	0.655	0.93	6.15
Ludwigia peruviana	1.073	0.33	9.69
Morus alba	1.073	0.334	9.69
Nephrolepis cordifolia	1.073	0.433	9.69
Oxalis corniculata	1.348	0.161	11.88
Passiflora edulis	1.073	0.342	9.69
Rubus spp.	0.840	0.661	7.75
Senna pendula	1.073	0.334	9.69
Tradescantia fluminensis	1.181	0.276	10.56

**Table 5** List of independent DISTLM comparisons between weed species abundance

 and ant species composition.

\* Significant at  $\alpha = 0.05$ 

The ant species that responded most greatly to the proportion of weed cover (as indicated by a Spearman Rank Correlation Coefficient of >|0.5|) were *Pheidole* 003 (Spearman  $\rho = -0.73$ ), *Paratrechina* 002 (Spearman  $\rho = -0.53$ ), *Monomorium* 001 (Spearman  $\rho = 0.60$ ) and *Meranoplus* 001 (Spearman  $\rho = 0.66$ ). The response of these species to differences in weed cover across the study sites is displayed graphically in Figure 3.

# 4. Discussion

There were no significant differences in riparian ant assemblages between urban and rural catchments. This is counter to our hypothesis and somewhat surprising considering the magnitude of land use change observed. There are a number of reasons that may explain this result.

First, it may be that ants do not respond *per se* to catchment land use. If this is so, it differs from studies such as Thompson & McLachlan (2007), who identified a simplification of ant communities with greater urbanisation, and Ives et al. (2011a), who found that the proportion of road adjacent to riparian corridors in northern Sydney influenced significantly the species richness of ant communities and their composition. However, since the intensity of urbanisation influences the magnitude of ecological responses (Vepsalainen et al. 2008), the impacts of urban land use in this study may not have been sufficiently greater than those of the preexisting rural land use to affect significant changes in riparian ant communities.

It is possible therefore that the failure to observe differences in ant assemblages is due to the presence of a disturbance threshold that both the urban and rural catchments have already crossed. Just as aquatic macroinvertebrate communities respond most dramatically to changes in impervious surface of <10% (Walsh et al. 2005a) and undergo significant stress when agriculture increases past ~30 % (Quinn and Hickey 1990), clearing of vegetation for agriculture may have already induced substantial change to riparian communities. Additional urbanisation may therefore have resulted in only marginal additional effects on ant communities. Ecological thresholds have been discovered in studies of urban-rural gradients for a range of taxa. For example, Blair and Launer (1999) observed a loss of butterfly species from habitats experiencing even minor human perturbation such as low levels of grazing or recreational use. Burton and Samuelson (2008) also failed to distinguish significant differences in native vegetation stem density between urban and agricultural lands (both environs were sparser than natural areas). Similarly, although Moffatt et al. (2004) identified strong compositional turnover in riparian vascular plant assemblages across a rural-urban gradient, they failed to observe a significant difference in understory diversity between suburban and rural forests. While these studies indicate that the thresholds for floristic structural change are likely to lie between natural and rural landuses, and between suburban and high-density urban areas, Sadler et al. (2006) found that the urban-rural gradient observed for carabid beetle assemblages was driven largely by differences between rural and suburban habitats.

Second, ants may respond to land use change but at a different spatial or temporal

scale to that which we measured. Arthropods are influenced strongly by microhabitat changes (Lassau and Hochuli 2004; Pik et al. 2002) and may not respond directly to broad scale land-use change in a surrounding catchment. This idea is supported by the fact that the only two environmental variables found to influence ant diversity were characteristics of the understorey, namely the height and proportion of weed species (Table 3). Ant communities may therefore be affected much more by *local* environmental factors than catchment-scale ones. Furthermore as species extinctions in habitat fragments can continue long after isolation (Hahs et al. 2009) and changes to biodiversity can occur for many decades post exurban development (Hansen et al. 2005), it is possible that the riparian ant assemblages we sampled from urban catchments may become increasingly differentiated from those in rural catchments over longer time periods.

Third, ants may be ecologically disconnected from the adjacent stream that has been altered physically and chemically as a result of the catchment land use. This idea is supported by Paetzold et al. (2005) who studied the diet of terrestrial arthropods in riparian zones using stable isotope analysis. They found that unlike other invertebrate taxa, ants derive very little of their diet from aquatic organisms. Although ants responded numerically to artificial enrichment of aquatic insects in their study, they displayed no response to a decrease in this food source, suggesting that aquatic measures of environmental quality do not relate to factors affecting ant assemblages.

Another source of disconnection between catchment land use and riparian ant communities may be through the riparian vegetation itself. While 'matrix' habitat

within human-dominated areas has been found to affect ant assemblages in urban habitat remnants (Gibb and Hochuli 2002; Lessard and Buddle 2005; Thompson and McLachlan 2007; Yamaguchi 2004), the wide riparian corridors in this study may have buffered streamside habitats from stressors derived from the urban/rural landscape. Anthropogenic pressures buffered by riparian corridors may include litter and chemical pollution from roads, agricultural chemicals, invasive species and physical trampling (see Fennessy and Cronk 1997; Norris 1993; Wenger 1999). Ives et al. (2011b) found a significant relationship between riparian corridor width and the composition of ant communities, however further research on the magnitude and penetration distance of urban edge effects would greatly aid the setting of appropriate corridor widths for the protection of riparian communities from surrounding land use (see Davies et al. 2011).

#### Weeds and Ant Community Assemblages

The small number of significant correlations between measures of ant diversity or composition and vegetation and water quality variables (Tables 3 and 4) is consistent with the findings by Uno et al. (2010) who failed to identify relationships between ant species richness and site environmental factors in Detroit and Toledo, USA. However, the strong association between riparian ant assemblage structure and the dominance and composition of weeds at a site (Table 4) is noteworthy from both ant ecology and riparian management perspectives.

Plants influence ant assemblages primarily through (a) affecting abiotic conditions of their surrounding environment (e.g. temperature, shading, soil composition and

structure), which influence nest establishment, and (b) influencing the type and abundance of food, which affects ant behavior, colony growth and the competitive ability of different species (Blüthgen and Feldhaar 2010). Furthermore, there is evidence that plant species richness can promote arthropod species richness through provision of a greater variety of resources (Siemann et al. 1998). However, the fact that only weed composition influenced ant communities points to differences in traits and function between exotic and native plant species as being of central importance to invertebrate communities. Weeds are commonly opportunistic or ruderal species that easily establish areas of high disturbance, light and space features common in riparian zones (Tickner et al. 2001). Traits such as these combined with release from herbivores and an ability to thrive in nutrient-rich environments (uncommon in the Australian context) gives them a competitive advantage (Lake and Leishman 2004). This can result in severe alteration of the structure and composition of riparian vegetation, which in turn can affect the hydrology and morphology of streams (Zavaleta et al. 2001). In addition, wholesale changes in vegetation composition that often result from weed invasion can have significant impacts on the environmental availability of nutrients (Richardson et al. 2007). This can influence ant community structure as a function of the nutrient requirements of different species (Blüthgen and Feldhaar 2010).

It appears that because ants respond most greatly to total weed cover that the degree of habitat structural change associated with invasion is of greater importance to ants than whether a particular plant species is indigenous or non-indigenous. Significant ecological impacts can often be attributed to a small number of dominant

alien species (Holmes et al. 2005). The fact that *Ligustrum sinense* and *Ligustrum lucidum* were both highly correlated with total weed cover and the two species most strongly associated with ant composition (Table 5) supports this assertion. *Ligustrum spp.* easily colonise riparian environments due to their prolific release of berries, and can quickly dominate riparian vegetation and suppress native flora (Burton and Samuelson 2008; Merriam and Feil 2002). Shading often increases with weed invasion and has been found to be a significant driver of ant community modification (Andersen and Majer 2004; Brian and Brian 1951). The impact of weeds on microhabitat appears therefore to be a primary driver of ant assemblage structure, and may even mask the expression of other landscape impacts (see Hooper et al. 2005).

#### What Drives Weed Invasion?

Riparian zones are naturally susceptible to weed invasion because of their linear structure, high levels of disturbance and because propagules are easily transported via flowing water (Brown and Peet 2003; Burton and Samuelson 2008; DeFerrari and Naiman 1994; Johansson et al. 1996; Planty-Tabacchi et al. 1996). Riparian invasibility can vary as a function of catchment scale human activity and weed density (Naiman et al. 2005; Vidra and Shear 2008), geomorphic style of the stream (Hupp and Osterkamp 1996; Tickner et al. 2001), and small scale habitat characteristics such as channel morphology or localised human perturbations (Richardson et al. 2007; Tickner et al. 2001). However, even with a low proportion of invasive species, disturbance events such as flooding can function over time as a catalyst for wholesale weed invasion if a sufficient supply of propagules is present

and the invasive species has the ability to rapidly occupy new habitats (Richardson et al. 2007). The fact that the degree of weed invasion can quickly increase once a density threshold has been crossed (Buckley et al. 2007) may explain why the study sites displayed such variability in weed dominance.

#### Ant Species that are Influenced by Weeds

The species of ants that responded most greatly to variation in weed cover potentially provide insight into the species traits that are favoured by weed-dominated riparian environments. However, the application of Andersen's functional group approach to classification of ant genera (Andersen 1995a) does not reveal consistent ant trait patterns at our sites. Monomorium1 responded positively to weed cover and *Pheidole3* negatively, yet both are classed as Generalised Myrmicinae. Similarly, abundances of Paractrechina2 declined with increasing weed cover yet the genus belongs to the Opportunists functional group, typically associated with disturbed and/or weed dominated environments (Gibb and Hochuli 2002; Hoffmann and Andersen 2003). The genus Meranoplus is not placed in any functional group (Andersen 1995a) yet increased strongly in abundance with weed cover. These results serve to highlight the need for further research into ant classification schemes specifically designed for urban environments at small scales (Andersen 1997). Nevertheless, given that the genera mentioned above derive a significant component of their diet from seeds, elaeosomes and/or extrafloral nectaries (Rico-Gray and Oliveira 2007), they may be affected in various ways by the transition in vegetation composition resulting from weed invasion.

#### Management Implications

These findings have practical implications for the management of riparian corridors and for how land use change can be mitigated to minimise ecological impacts on ecological communities. First, the lack of significant differences in ant assemblages between urban and rural catchments highlights the complexity of terrestrial macroinvertebrate communities and the high degree of biotic turnover they express over a range of spatial scales (Hendrickx et al. 2007). This should serve as a word of caution to policymakers, as macro-invertebrate communities may not observe the land use management boundaries typically imposed on the landscape. Instead, if invertebrate diversity is to be considered in management strategies, managers must recognise that these animals often respond to environmental conditions at scales smaller than those traditionally included in management plans for mammals, birds and reptiles (Andersen et al. 2004). This has implications both for selecting appropriate sizes of protected habitat patches and the management regimes that are applied to them.

The significant influence of weed cover on ant communities suggests that managing invasive species should be a major priority in the management of catchments for biodiversity. Indeed, Pacheco and Vasconcelos (2007) conclude from their study of ants in urban habitats in Brazil that the presence of native vegetation within urban parks is important for maintaining a diverse ant fauna. However, an understanding of the specific processes that have contributed to invasion and the scales at which they operate is critical to ensure weed management is efficient and effective. For example, attempts to restore native vegetation along a reach while stands of alien plants

remain upstream and catchment fluvial dynamics are severely altered will most likely be unsuccessful (Richardson et al. 2007). Furthermore, even when best practice strategies are adopted, management of weeds can be costly and require substantial commitment over time because of the need to manage regrowth and revegetate sites with native species (Abell 2005; Vidra and Shear 2008).

# Conclusion

This study demonstrated that the factors that influence riparian ant assemblages are complex and potentially operate at a range of different spatial scales. Correlations between ant communities and floristic characteristics, especially weed dominance, suggest that ants respond more greatly to local scale environmental conditions and biological interactions than catchment scale land use. However, because this study did not consider catchments in natural condition, the possibility remains that both the urban and rural sites that were assessed had already transitioned past an ecological threshold induced by land use transformation. We recommend that environmental managers should control riparian weeds regardless of catchment context in order to promote the broader ecological health of riparian corridors. Although significant differences in ant communities were not observed between urban and rural land uses, catchment scale urbanisation of greenfield sites is likely to have additional farreaching environmental impacts not addressed by this paper, and should not necessarily be encouraged based on the results of this study.

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Camponotus1	N	-	(0.693)	0	(0)	0	0)	0	(0)	0	(0)	0	(0)	<u> </u>	(0.717)	0	(0)	0	0)	0	0	0	(0)	0	0
Heteroponera1	2	0	(0)	0	(0)	2	(1.382)	N	(1.408)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Monomorium3	N	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.688)	0	(0)	0	(0)	0	(0)	0	(0)	ω	(2.166)
Myrmecia2	N	0	(0)	-	(0.712)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.705)	0	(0)	0	(0)	0	(0)	0	(0)
Plagiolepis1	N	6	(4.074)	0	(0)	0	(0)	-	(0.715)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Prolasius5	N	0	(0)	0	(0)	-	(0.675)	0	(0)	N	(1.394)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Strumigenys2	2	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)		(0.702)	0	(0)	0	(0)	0	(0)	-	(0.679)	0	(0)
Camponotus2	-	-	(0.731)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Cryptopone1	-	0	(0)	0	(0)	ω	(2.076)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Dolichoderus1	-	-	(0.7)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Hypoponera1	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.787)	0	(0)	0	(0)	0	(0)
Hypoponera2		0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.793)	0	(0)	0	(0)	0	(0)
Hypoponera3	-	1	(0.705)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Hypoponera4	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.716)
Lordomyrma1	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.708)	0	(0)	0	(0)	0	(0)	0	(0)
Melophorus1	-	0	(0)	-	(0.712)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Monomorium4	-	0	(0)	0	(0)	0	(0)	-	(0.71)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Notoncus2	<u> </u>	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	ω	(2.01)	0	(0)	0	(0)	0	(0)	0	(0)
Oligomyrmex1	-	-	(0.679)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Pachycondyla1	-	0	(0)	0	(0)	-	(0.691)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Plagiolepis2	-	-	(0.679)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Ponera1	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.707)
Ponera2	-	0	(0)	0	(0)	0	(0)		(0.695)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Prionopelta1	-	0	(0)	0	(0)	-	(0.675)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Prolasius3		0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.721)	0	(0)	0	(0)	0	(0)	0	(0)
Pseudonotoncus	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.704)	0	(0)	0	(0)	0	(0)	0	(0)
Solenopsis4	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)		(0.713)	0	(0)
Solenopsis5	-	-	(0.679)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)
Solenopsis6	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(1)	0	(0)	0	(0)
Tapinoma	-	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.741)	0	(0)	0	(0)	0	(0)	0	(0)
Tetramorium1		-	(0.677)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)

abundance	es adjuste	d t	or / sa	Imp	ples at	ter	rareta	acti	on.																
Morphospecies	No. of Sites		포		R2		R3		R4		R5		R6		Ē		U2		U3	c	4		U5		<b>U</b> 6
Anonychomyrma1	12	52	(36.231)	32	(22.6)	118	(82.322)	77	(53.526)	646	(456.552)	225	(157.768)	46	(32.253)	72	(50.148)	39	(30.934)	230	(230)	226	(156.339)	ω	(5.59)
Pheidole1	12	485	(334.251)	137	(95.977)	60	(42.238)	214	(147.213)	230	(159.989)	ഒ	(44.319)	-	(0.714)	535	(369.014)	241	(187.158)	-	(1)	-	(0.698)	93	(65.682)
Aphaenogaster1	11	39	(27.076)	15	(10.434)	-	(0.709)	-	(0.719)	თ	(3.507)	16	(11.288)	1	(7.72)	14	(9.861)	0	(0)	ω	(3)	-	(0.71)	4	(2.785)
Solenopsis1&3	=	œ	(5.453)	N	(1.412)	თ	(3.474)	15	(10.572)	-	(0.71)	ω	(2.121)	12	(8.397)	36	(24.364)	14	(10.829)	-	(1)	0	(0)	26	(18.188)
Pheidole2	10	0	(0)	16	(11.155)	11	(7.569)	21	(14.86)	ഗ	(3.576)	0	(0)	25	(17.536)	8	(5.591)	7	(5.469)	22	(22)	12	(8.485)	36	(24.986)
Prolasius2	œ	4	(2.862)	21	(14.832)	4	(2.837)	0	(0)	0	(0)	7	(4.902)	ω	(2.132)	7	(4.957)	7	(5.404)	-	(1)	0	(0)	0	(0)
Rhytidoponera1	ω	0	(0)	6	(4.26)	112	(78.036)	47	(32.627)	0	(0)	0	(0)	0	(0)	10	(7.188)	-	(0.787)	⊐	(11)	-	(0.687)	94	(66.435)
Paratrechina1	7	6	(4.31)	0	(0)	0	(0)	0	(0)	N	(1.436)	0	(0)	7	(4.872)	0	(0)	N	(1.552)	N	(2)	N	(1.374)	-	(0.69)
Crematogaster1	6	ഗ	(3.462)	N	(1.427)	0	(0)	-	(0.678)	0	(0)	N	(1.408)	0	(0)	0	(0)	ω	(2.338)	-	Ξ	0	(0)	0	(0)
Leptomyrmex1	6	ω	(2.131)	0	(0)	N	(1.361)	ი	(4.239)	0	(0)	0	(0)	0	(0)	-	(0.67)	0	(0)	N	(2)	N	(1.374)	0	(0)
Solenopsis2	6	1	(7.951)	0	(0)	N	(1.406)	0	(0)	ω	(2.13)	0	(0)	-	(0.7)	0	(0)	0	(0)	0	0	-	(0.679)	N	(1.4)
Meranoplus1	σ	0	(0)	0	(0)	9	(6.361)	0	(0)	0	(0)	0	(0)	0	(0)	19	(12.718)	19	(14.577)	0	0	ω	(5.498)	41	(29.202)
Monomorium1	σı	0	(0)	-	(0.7)	0	(0)	-	(0.719)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.762)	0	0	-	(0.698)	N	(1.375)
Paratrechina2	σı	œ	(5.772)	-	(0.701)	-	(0.691)	N	(1.438)	0	(0)	0	(0)	-	(0.698)	0	(0)	0	(0)	0	0	0	(0)	0	(0)
Paratrechina3	σı	-	(0.7)	4	(2.83)	-	(0.701)	<b>_</b>	(0.678)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	0	0	(0)	-	(0.69)
Pheidole3	σı	111	(77.765)	0	(0)	24	(16.56)	0	(0)	ω	(2.154)	49	(34.571)	0	(0)	0	(0)	0	(0)	10	(10)	0	(0)	0	(0)
Prolasius1	ы	-	(0.704)	4	(2.784)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.704)	9	(6.03)	0	(0)	-	Ξ	0	(0)	0	(0)
Prolasius6	σı	0	(0)	6	(4.203)	-	(0.695)	0	(0)	0	(0)	0	(0)	0	(0)	18	(12.684)	0	(0)	N	(2)	0	(0)	-	(0.716)
Strumigenys1	σı	N	(1.377)	0	(0)	0	(0)	N	(1.356)	7	(5.029)	0	(0)	-	(0.7)	0	(0)	ω	(2.31)	0	0	0	(0)	0	(0)
Amplyopone1	4	0	(0)	0	(0)	-	(0.701)	ω	(2.139)	-	(0.693)	0	(0)	-	(0.714)	0	(0)	0	(0)	0	0	0	(0)	0	(0)
Notoncus1	4	-	(0.679)	თ	(3.457)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.717)	N	(1.416)	0	(0)	0	0	0	(0)	0	(0)
Rhytidoponera2	4	0	(0)	0	(0)	4	(2.82)	N	(1.43)	0	(0)	0	(0)	25	(17.543)	0	(0)	0	(0)	0	0	-	(0.679)	0	(0)
lridomyrmex1	ω	-	(0.7)	13	(9.253)	-	(0.691)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	0	0	(0)	0	(0)
Linepithema1	ω	0	(0)	-	(0.7)	0	(0)	0	(0)	ω	(2.122)	0	(0)	0	(0)	-	(0.721)	0	(0)	0	0	0	(0)	0	(0)
Monomorium2	ω	-	(0.694)	0	(0)	0	(0)	N	(1.379)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	0	-	(0.708)	0	(0)
Myrmecia1	ω	0	(0)	0	(0)	-	(0.694)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.685)	0	(0)	0	0	0	(0)	8	(5.456)
Prolasius4	ω	N	(1.431)	ω	(2.124)	0	(0)	0	(0)	N	(1.414)	0	(0)	0	(0)	0	(0)	0	(0)	0	0	0	(0)	0	(0)
Acropyga1	N	0	(0)	0	(0)	6	(4.25)	4	(2.86)	0	(0)	0	(0)	0	(0)	0	(0)	0	(0)	0	0	0	(0)	0	(0)
Anillomyrma1	2	0	(0)	-	(0.712)	0	(0)	0	(0)	0	(0)	0	(0)	-	(0.688)	0	(0)	0	(0)	0	0	0	(0)	0	(0)

Chapter 5 Riparian Ants in Rural and Urban Catchments

# **CHAPTER 6**

**General Discussion** 

The goal of this thesis was to understand the ecological health and biodiversity patterns of urban riparian corridors in order to inform public policy and direct the best practice management of these systems. The results outlined in the previous chapters have provided insight to this end and are synthesised in this final discussion section. As outlined in Chapter 1, the first aim of this thesis was to explore the role of science in the formulation and implementation of local environmental policy instruments (EPIs), and to provide a framework for the integration of biodiversity concepts into such policies.

This first aim was achieved in Chapter 2, in which environmental legislation was reviewed at international, national, state and local levels. The scientific pursuit of defining and assessing biological diversity was also summarised and integrated into a framework that could allow local environmental practitioners to grasp and apply the concepts of biodiversity science to local contexts. It was argued in Paper 1 of Chapter 2 that such a holistic framework is necessary for local environmental managers to achieve biodiversity conservation goals that are true to the principles of Ecologically Sustainable Development (ESD). The importance of integrating science in environmental policy was emphasised further in the review and critique of NSW riparian policy frameworks (Chapter 2, Paper 2). In that paper, significant policy reform was proposed in recognition of the unique form and function of urban rivers and riparian zones and the factors that influence them.

The second aim of this thesis was to investigate the relationship between riparian corridor width and biodiversity in an urban setting. This aim was selected because of
the significance of corridor width in local riparian policy, and was addressed in Chapter 3. Through assessment of ant and plant communities in the Ku-ring-gai Local Government Area (LGA), corridor width was not significantly related to the species richness of these groups, but corridor width did influence ant and plant assemblage composition. In particular, corridor width influenced strongly the proportion of opportunistic and dominant ant species, and the proportion of weeds within the riparian corridor.

The third aim of this thesis was to identify the principal environmental factors affecting riparian biodiversity. This was addressed in Chapter 4. This chapter arose from a desire to explore the sources of unexplained variability in biological data observed in Chapter 3. Interestingly, despite the relationship between biodiversity and corridor width identified in Chapter 3, corridor width was not related to biological datasets as closely as other environmental factors. Minimum adequate models did not contain corridor width as a predictive variable of either ant diversity or ant composition. While plant composition appeared to be affected by corridor width, this was most likely due to the inclusion of novel communities upslope in wider corridors rather than any mediating effect of a vegetated buffer on floristic assemblages.

The incomplete account of between-site variance in ant assemblages by environmental variables, combined with the significant influence of adjacent roads in Chapter 4, led to the more targeted investigation of the effect of land use on riparian ants presented in Chapter 5. Contrary to predictions, no significant effects of land use on riparian ant diversity or composition were found in study sites from north-west

Sydney. Instead, ants were influenced more strongly by local vegetation structure and the dominance of exotic vegetation within streamside habitats.

Through synthesis of the results presented in earlier chapters, this discussion will address firstly how this thesis contributes to a general understanding of biodiversity patterns and ecological function in urban riparian corridors. Since the application of empirical science to policy and ecological science *per* se have been of equal focus in this thesis, the second part of this discussion identifies key recommendations for the management of urban riparian corridors. Finally, future directions in urban ecological research and biodiversity management are discussed.

# 1. The ecological functioning of urban riparian zones

This thesis has identified that biodiversity patterns in urban riparian corridors are complex, respond to a range of environmental influences, and vary between sites and taxonomic groups. Based upon the research in this thesis, a model of the key components and forces affecting the biodiversity of riparian corridors is presented in Figure 1.



**Figure 1:** Flow chart of the dominant biophysical elements and processes that affect the biodiversity of urban riparian corridors. Arrow width indicates the relative scale of influence of factors on components/processes. Specific components of the diagram are discussed below. Note that the term "reserve configuration" relates to a range of attributes of riparian corridor design such as width, connectivity, area and shape.

As outlined in Chapters 3 and 4, the diversity of urban riparian corridors (the bottom box in Figure 1) is a combination of the biotic communities adjacent to the stream (i.e. the 'streamside communities' box in Figure 1), and those occurring upslope away from the direct influence of the watercourse (i.e. Figure 1, 'hillslope communities' box). These communities are comprised of ants, plants and a range of other organisms from microbes to vertebrates, which are likely to influence ant and plant communities through interspecific interactions (as depicted by the arrows between ovals within the 'streamside communities' and 'hillslope communities' boxes in Figure 1). Chapters 3, 4 and 5 consistently highlighted the importance of weeds in affecting the diversity of ants and plants. The box labelled "weeds" in Figure 1 has thus been positioned centrally, and is shown to be influenced by landscape-scale and localscale factors as well as human actions such as bush regeneration (as demonstrated by in Chapters 3 to 5). The effects of environmental variables on ant and plant communities (identified in Chapter 4), and the influence of weeds and local habitat structure (described in Chapter 5) are represented by the series of arrows stemming from the top boxes. Importantly, the influence of urban edge effects is much more significant for hillslope communities than streamside communities because of their proximity to the urban matrix. However, the degree to which these communities experience these edge impacts is related to the width of the riparian corridor, which functions as a buffer (see Chapter 3), as is shown by the text on the left hand side of Figure 1.

A number of the processes presented in Figure 1 are discussed in further detail below. These include (1) the influence of riparian corridor width, (2) landscape and local environmental factors, (3) the impact of weeds, and (4) biological interactions. Other potential sources of variability in biodiversity that may be pertinent to urban riparian corridors but are not included in Figure 1 are also discussed below.

## 1.1 Corridor Width

Results from Chapters 3 and 4 revealed that ant and plant communities respond in complex and non-linear ways to variation in corridor width. Numerous mechanisms combine to give rise to the significant relationships observed between corridor width and biodiversity (see Chapter 3), despite the fact that the importance of corridor width is diminished in the context of other environmental variables (see Chapter 4). A number of these mechanisms are outlined below.

The most significant way in which corridor width affects the biodiversity of riparian corridors is through influencing the proportion of streamside to hillslope habitat (Chapters 3 and 4). As biological communities can express a high degree of beta diversity (turnover) along this environmental gradient (Naiman and Decamps 1997), the total biodiversity of a riparian corridor will be influenced significantly by its width. This phenomenon is denoted by the large and small arrows at the bottom right of Figure 1.

When riparian biological communities are considered apart from the streambankhillslope gradient (i.e. through studying streamside habitats alone, as seen in Chapters 3 and 4), corridor width has a significant, albeit mild influence on biodiversity. Mechanisms for this trend include the species-area relationship (Connor and McCoy 1979), the intermediate disturbance hypothesis (Connell 1978) and edge influence (Ries et al. 2004) (see the "urban edge influence" arrow in Figure 1). Corridor width *per se* does not directly influence riparian biodiversity. Rather, the width of a riparian corridor modifies the degree of influence exerted by other environmental factors and ecological processes.

## 1.2. Environmental determinants of biodiversity in urban riparian systems

The question of what principally drives biodiversity in urban riparian zones is at the heart of this thesis. Although the effect of corridor width on the diversity of ants and plants was significant, Chapter 4 demonstrated that other environmental factors play a more important role (see Tables 2 and 3, Chapter 4). This was supported by the observation that substantial compositional variation in ant assemblages exists between sites, despite corridors being of similar widths (Chapter 5). The influence of these environmental factors is represented in Figure 1 by the by the ovals within the top two boxes entitled 'landscape scale' and 'local scale'. The number of pathways by which environmental factors affect riparian biological communities as indicated by the arrows help to explain the complexity of the results observed in Chapter 4. For example ant species richness is affected by a combination of local factors (i.e. % adjacent road, perimeter:area ratio and riparian connectivity) (see Chapter 4, Table 3). Although in Figure 1 these factors are split between two spatial scales, in reality each factor varies uniquely over space.

While many environmental factors were considered in Chapters 4 and 5, there remains the possibility that abiotic variables that were not measured exert a greater influence on riparian ant and plant biodiversity than those that have been described.

Natural conditions such as sunlight levels or the water holding capacity of the soil, or anthropogenic stressors such as pollutants or trampling are potential influences. However, given the number of variables that were considered in Chapters 4 and 5, it is unlikely that another additional environmental variable would alone explain the large amount of variation in the biological datasets. Rather, the finding that ants and plants are related strongly to the dominance of weeds at both the Ku-ring-gai and north-west study locations (Chapters 3, 4 and 5) indicates that biological change may also be critical in influencing riparian biodiversity.

### 1.3. Weeds

The relationships between weed abundance and biotic assemblages (Chapters 3, 4 and 5) is a consistent finding of this thesis. Weeds substantially degrade ecosystem function and threaten biodiversity (Vitousek 1990), and as such are a major focus of ecological management in Australia (Downey et al. 2009). Although this thesis was not structured to study specifically the influence of weeds on riparian communities, the apparent importance of weeds to riparian ecosystems warrants special attention and suggests that their control is a critical aspect for successful riparian management.

The weed component of plant assemblages was the principal driver of the relationship between corridor width and vegetation composition (Figure 5 in Chapter 3; Chapter 4). However, this finding was primarily because hillslope communities are less affected by weed invasion as a function of their disconnection from stream

processes, and yet they comprise a greater proportional area in wider riparian corridors (Chapters 3 and 4). The higher invasibility of streamside communities as a function of nutrient enrichment and periodic physical disturbance is well recognised globally (Brown and Peet 2003; DeFerrari and Naiman 1994), and Sydney is no exception (King and Buckney 2002; Lake and Leishman 2004). Thus from a weed management perspective, the factors that control invasion within this naturally susceptible area is of great importance.

Soil pH was the principal driver of native plant composition in streamside habitats in Ku-ring-gai (Chapter 4), while the proportion of exotic species relative to natives was influenced significantly by catchment land use in north-west Sydney (Chapter 5). Soil pH varies with soil type (i.e. sandstone soils are more acidic than shale-derived soils), and soil type is also related to nutrient levels (i.e. shale soils typically contain higher levels of phosphorus than sandstone). Furthermore, while soil nutrients were not tested explicitly in this thesis, previous research has identified nutrient enrichment as an important factor driving weed invasion in Sydney (e.g. Lake and Leishman 2004).

The ecological mechanisms that give rise to compositional change in riparian vegetation can be inferred from the results herein and from the literature. Lake and Leishman (2004) measured morphological traits of native and exotic species in Sydney bushland and found that invasive species have consistently higher specific leaf-area ratios and softer leaves, larger seeds and greater capacity for vegetative propagation. Interestingly, many of these traits such as large leaf size are also

shared by native riparian species such as Coachwood (*Ceratapetalum apetalum*), but not by sclerophyllous hillslope vegetation. Therefore, rather than nutrient enrichment resulting in native riparian plant mortality as has been observed in sclerophyllous sandstone vegetation (Groves and Keratis 1976), it is through biological competition that compositional change is more likely to occur, facilitated by physical disturbance from the stream (discussed in Chapters 3 and 5).

Recent experimental research by MacDougall and Turkington (2005) explored the question of whether weed invasion in disturbed environments is driven primarily by environmental disturbance or biotic factors (i.e. exotics outcompeting natives) and concluded that both processes appear to operate in concert. The inverse relationship observed between exotic and native riparian plant species richness in Ku-ring-gai (Figure 6, Chapter 3) provides support for MacDougall and Turkington's (2005) assertion of the importance of competitive exclusion in weed invasion. Similar relationships between native and exotic species richness have been observed elsewhere in Sydney (e.g. Williams and Roberts 2005). Furthermore, the effect of catchment land use on the proportion of exotic species in riparian zones in north-west Sydney (Chapter 5) also suggests that environmental disturbance plays a role in influencing weed invasion. The greater dominance of riparian weeds in urban catchments with a large proportion of impervious surfaces compared to those with less impervious surfaces is related to the greater and more frequent physical disturbance from flood events in urban catchments (Paul and Meyer 2001). With each periodic inundation, exotic species that possess superior propagule production and seedling establishment rates (relative to native species) are readily established

within the riparian zone and this can quickly result in wholesale modification of the vegetation community (Richardson et al. 2007). The greater frequency of flood events in urban catchments is likely to be most significant in structuring riparian communities located directly adjacent to the stream. The relative influence of the flashy urban hydrologic regime would decline with distance upslope, as the sealing of surfaces in urban catchments increases the frequency of low to medium overbank flows more than large floods.

Weeds not only influence vegetation composition, but can also affect other organisms within the same habitat. Their influence on microhabitat, soil organic matter and food resources are known to affect ant communities significantly (discussed in Chapters 3, 4 and 5). The influence of weeds on ant communities was demonstrated clearly in Chapter 5 (see Figure 3), with the physical domination of weeds on a site's physical structure being of significant importance. This result is supported by other studies such as that by Grimbacher & Hughes (2002), who identified significantly greater abundances of Opportunistic ants in plots with a high percentage of weed cover compared to a greater proportion of Dominant Dolichoderines at reference sites with low weed cover.

It is possible, however, that changes in ant community composition precede floristic changes. The presence of different ant species could alter seed dispersion or soil conditions, which in turn may favour different species of vegetation. However, Lake and Leishman (2004) found that no exotic plants possessed seeds adapted to dispersion by ants compared with 42% of native species. This suggests therefore

that ant compositional change observed in this thesis is more likely to be in response to initial floristic modification, rather than vegetation differences reflecting preceding alterations in ant assemblage composition.

## 1.4. Biological interactions

Given the proven effect of weeds on riparian biological communities, it is likely that other biological interactions are also important in determining the biodiversity of urban riparian corridors. Although this thesis did not explicitly measure biological interactions, ecological theory and empirical studies of interspecific relationships can be used to infer processes that are likely to operate in urban riparian systems. These biological interactions are denoted in Figure 1 by the arrows between the ovals labelled 'ants', 'plants' and 'other organisms' (dashed lines indicate elements that were not addressed specifically in this thesis). Before these interactions are explored, the relative importance of environmental and biological factors in determining the composition of biological communities are discussed.

The degree to which species composition at a set locality is determined by environmental conditions or the historical order of species migration has been the subject of much debate (Chase 2003). While environmental conditions may predispose a site to the presence of certain species, in many cases it is overly simplistic to treat species composition as a simple mathematical function of certain environmental characteristics (Kirkpatrick and Brown 1994). Observation of contrasting assemblages of species in habitats with similar environmental conditions

suggests that the original composition of species in a particular habitat is likely to influence future community development (e.g. Drake 1991). However, this phenomenon is more likely to be observed in areas with a large regional species pool, low ecological connectivity, high productivity and low disturbance, while areas with a small regional species pool, high ecological connectivity, low productivity and high disturbance may respond more predictably to environmental conditions (Chase 2003). Whether urban riparian ecological systems align more closely with models of 'single stable equilibrium' (i.e. species assemblage is the product of environmental conditions) or 'multiple stable equilibria' (i.e. many community assemblages are possible for the same environmental conditions) is not yet known (see Chase 2003).

Measuring and understanding the nature of species interactions is very time and labour intensive even for two species, let alone entire communities of organisms, and therefore this research has been underrepresented in studies of biodiversity (Del-Claro 2004; Ollerton and Cranmer 2002). However, in recognition of the importance of species interactions in ecosystem function, recent research has looked at quantifying 'interaction diversity', which can be used in similar ways to more established diversity metrics (Dyer et al. 2010). Exploration of these interactions can help illuminate biodiversity patterns, since the introduction of a new species can result in wholesale ecosystem transformation (e.g. the impact of the introduced Argentine ant on native ant populations: Suarez et al. 1998). Species interactions that are likely to be significant in the riparian zones sampled in this thesis are explored briefly below.

#### 1.4.1. Ant – Ant Interactions

Competition can be defined as "a negative interaction between individuals depending on a shared, limited resource" (Parr and Gibb 2010, p. 77) and has been described as the hallmark of ant ecology (Hölldobler and Wilson 1990) because of its crucial role in structuring assemblages across a range of habitats. Research on ant competition has centred largely around the degree of dominance exerted by certain species on other ants and has formed the basis for functional group analyses (Andersen, 1995) as utilised in Chapters 3 to 5. The nature of these 'dominance hierarchies' was first discussed by Wilson (1971). They have since been found to affect aspects of assemblage structure and function, such as species richness and the ability to exploit and defend food resources (see Parr 2008; Savolainen and Vepsäläinen 1988). Mechanisms that influence ant community structure via interspecific competition include:

- Differences in foraging behaviour: while dominant ants fiercely defend and monopolise food resources, they are often slower to discover them than opportunists (Fellers 1987).
- Differences in food sources: While most ants are omnivores, some are known to specialise on other insects or specific plants (Lach et al. 2010). For example, Formicinae (e.g. *Camponotus* spp.) contain the highest proportion of plant food in their diet, Dolichoderinae have intermediate levels, and Myrmecinae (e.g. *Pheidole spp.*) and Ponerinae are almost entirely carnivorous (Fiedler et al. 2007).

- Environmental stress: A negative relationship appears to exist between stress and competition. The ability of certain species to dominate an assemblage can be hampered under stressful conditions such as lower temperatures often associated with dense vegetation cover (Parr and Gibb 2010).
- Habitat complexity: The ability of behaviourally dominant species to locate, defend and monopolise food resources can be restricted in complex habitats (Gibb and Parr 2010). This is because large body and leg size common to dominant species can impede movement (Gibb 2005; Kaspari and Weiser 1999). Smaller, subordinate species can also escape more easily or avoid confrontation in complex habitats and therefore proliferate alongside behaviourly dominant species (Savolainen et al. 1989).
- The dominance-richness relationship: unimodal species richness responses to the abundance of dominant ants have been observed through assessment of ant behaviour at baits (Andersen 1992). This is due to a competitive exclusion threshold whereby dominant ants restrict subordinate species access once dominant species attain a certain density.

There is evidence of interspecific competition in ant communities at both the Ku-ringgai and north-west study sites surveyed in this thesis. For example, analysis of ant populations revealed a transition from high abundances of opportunist species in narrow corridors to dominant species in wide corridors. These species assemblages point towards the suppression of subordinate ants by *Anonychomyrma* and *Iridomyrmex* species in corridors wider than 60 m (Chapter 3). Furthermore, the relatively even distribution of functional groups in the middle 35-50 m category provides some support for the dominance-richness relationship mentioned above, and may help to explain the lack of a significant correlation between corridor width and ant species richness (Chapter 3, Table 1). The influence of *Pheidole megacephala* on ant assemblage composition in the study of wide riparian corridors in north-west Sydney (Chapter 5, Figure 3) also indicates that interspecific competition plays a significant role in structuring riparian ant communities, possibly through this species' ability to proliferate quickly and displace other ants (Hoffmann and Parr 2008). However, further research on the behavioural influence of this invasive species on ant communities in Sydney is needed to obtain a greater understanding of the nature of this relationship.

#### 1.4.2. Ant - Other Organism Interactions

## (a) Feeding

Ant communities are affected not only by ant-ant competition for food resources, but also by the presence and abundance of the organisms that provide their food. Seeds are collected by a small number of ant genera including *Monomorium* and *Pheidole* (observed at Ku-ring-gai and north-west Sydney study locations). Importantly, the types of seeds collected often differ according to ant body size (Traniello 1989). Consequently, a change in riparian vegetation composition as a result of urban development (observed in Chapter 5) could easily result in the loss of this resource for certain colonies. Interestingly, however, it appears that many species are able to alter their diet to respond to changes in food source availability (a function of habitat type) without altering the overall trophic structure of the community (Gibb and Cunningham 2011). As detailed research on the use of food resources by ant

assemblages is very much in its infancy, greater knowledge is needed to understand the extent to which degradation of riparian habitat through the input of toxins and pollutants or modification by weeds would impact the trophic dynamics of riparian ant assemblages.

#### (b) Mutualistic Relationships

Ants are known to interact mutualistically with a range of other organisms. Mutualisms have played an important role in the evolution of ants and are important in structuring many ant communities (Ness et al. 2010). For example, many ants provide protection to various insects in return for food such as those that fend off attackers in return for energy or nitrogen rich secretions from Hemiptera or Lepidoptera larvae (Fiedler 2001; Stadler and Dixon 2005). Of particular interest is that some species of *Anonychomyrma* (highly abundant in many sites in Ku-ring-gai) are associated in this way with some caterpillars (Lepidoptera) (CSIRO 2011).

Many plants also attract ants through the secretion of nectar from glands known as extra-floral nectaries in return for protection (Ness et al. 2010). Alternatively, others (particularly *Acacia* and *Grevillea* species) rely on ant-mediated seed dispersal through the attachment of an elaiosome to the seed as a food reward (Beattie 1985; Hughes and Westoby 1990; Rico-Gray and Oliveira 2007). Transformation of vegetation composition within riparian corridors in response to environmental stress or weed invasion is likely to result in significant changes to the assemblages of ants and possibly other organisms that rely on multualistic relationships with the indigenous vegetation.

## 1.5. Other sources of variation in riparian biodiversity

Two additional mechanisms exist that may contribute to variance in biodiversity between sites as seen in Chapters 3, 4 and 5. First, the observed diversity and composition of urban riparian corridors may not be a reflection of current environmental conditions, but of those that have existed previously. As discussed in Chapter 5, a lag period is associated commonly with the response of biological communities to landscape change such as habitat loss (Hahs et al. 2009; Helm et al. 2006). Likewise, it is possible that the effects of a reduction in corridor width or urbanisation in the areas adjacent to the sampled riparian sites are yet to be manifested fully in related ant and plant communities. In future research of urban ecological systems, it may be prudent therefore to consider the history of the landscape being sampled and undertake long-term studies. New land release areas on the outskirts of many cities are prime opportunities for this type of longitudinal study as the transformation of land use from rural to medium-density residential is typically rapid and extensive.

A second source of variability between sites may be that a naturally high degree of compositional change exists over relatively short geographical distances, in line with the observed "distance decay of similarity" phenomenon, otherwise known as spatial autocorrelation (Nekola and White 1999). Although this phenomenon is rooted in environmental differences (i.e. the further apart two points are in space, the more likely there will be a concomitant reduction in the similarity of environmental features), it is also brought about by limited dispersal of individuals (Soininen et al. 2007). Dispersal limitation is itself a function of (i) the existence of physical dispersal barriers

(current and historical) (e.g. valleys, ridges, habitat loss etc.) and, (ii) the inherent limitation of organisms to disperse over large distances even within physically homogeneous environments (c.f. Hubbell's (2001) Neutral Theory of Biodiversity) (Soininen et al. 2007). Assessment of ant and plant dispersal requires significant field effort and is beyond the scope of this study. Nevertheless, it is possible that the extant biodiversity of riparian corridors measured in this thesis has been influenced significantly by dispersal limitations rather than local environmental characteristics.

## 2. Urban riparian management and land use planning

## 2.1. Science-Policy Interaction

As contended in Paper 1 of Chapter 2 and in Appendix 1, effective environmental policy can only be achieved via a thorough scientific understanding of the pertinent issues and unique characteristics of a region. This thesis responds to this viewpoint by providing an in-depth exploration of the challenges of applying empirical science to real-world policy in a local context. The results from Chapters 3 to 5 are particularly challenging to integrate existing riparian policy frameworks because of the ecological complexity observed, the idiosyncratic nature of biological responses to individual environmental variables, and the degree of unexplained variation in statistical models.

Environmental policy operates most effectively when it is grounded in known, predictable and repeatable phenomena such that environmental responses to particular actions can be anticipated and predicted with a high degree of certainty (Bradshaw and Borchers 2000). This, however, is rarely the case when studying natural systems. Exceptions do exist such as the banning of CFCs in the Montreal Protocol or the phasing out of leaded petrol, the success of which can be attributed largely to the knowledge of the particular cause and effect relationships. However, many contentious or challenging environmental policies are a response to scientific complexities or uncertainties. Examples of this range from cutting greenhouse gas emissions to mitigate climate change to the promotion of 'sustainable' aquaculture (see New South Wales State Environmental Planning Policy No 62-Sustainable Aquaculture). It appears from the results presented in this thesis that protection of urban riparian systems through the current legislative and policy approaches will continue to be plaqued with contention because of the complexity of ecological patterns observed and the many competing priorities placed on riparian zones. Significant legal conflict over the protection of riparian corridors arises for two main reasons: (i) the economic value of land that could be developed is lost when it is granted protection as part of a riparian corridor, and (ii) uncertainty exists over the adequacy of riparian setbacks to achieve environmental functions, especially with regard to the space required for species habitat (see Figure 1 in Chapter 1). Indeed, this was the crux of the case Silverwater Estate Pty Ltd v Auburn Council [2001] in the NSW Land and Environment Court (NSWLEC 60) where a riparian corridor was required to function as habitat, an ecological conduit and a buffer for aquatic ecosystems. This and other examples pertinent to application of science to environmental policy in NSW are discussed in detail in Appendix 1.

The challenges encountered frequently when seeking to formulate and implement environmental policy stem broadly from the way communities typically understand the relationship between people and their environment. Environmental policies are birthed from political systems that begin from an anthropocentric worldview that perceive humanity as the centre, rather than those that view humanity as existing within the natural world (see Eckersley 1992). Current political systems are designed to ask 'how can human society be managed so as to function optimally and minimise its impact on the surrounding environment? (e.g. the National Strategy for Ecologically Sustainable Development (1992), the Water Act 2007 (Australian Government), or the newly released Commonwealth national urban policy, "Our Cities, Our Future"). Rarely do policies originate from pursuing the question of 'how do natural systems and processes respond to human presence and activities?'. The first question begins with understanding people; the second begins with understanding the broader environment and viewing people as components within a larger system. Full adoption of the second perspective would require wholesale change to governance structures, which may be unfeasible in the short term. Nevertheless, the simple act of adopting a greater understanding of natural systems as the basis for policy formulation will enhance the long-term efficacy of environmental policy significantly and move closer to attaining the goals of ESD. In any case, to not take this approach will continue to lead us down the same trajectory we have been for centuries; one of increased and accelerating environmental degradation and biodiversity loss.

While further research, practice and testing will in time reduce ecological uncertainty, there is a more immediate need to reconcile the current levels of uncertainty with robust environmental policy. This sentiment was expressed strongly by McClellan CJ in the case Stockland Development Pty Ltd v Manly Council [2004] NSWLEC 472 (discussed in Chapter 2). Indeed, the precautionary principle targets this very issue, namely through stating that a "when an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically" (Raffensperger and Tickner 1999; Science and Environmental Health Network 1998). This is an important concept as some level of uncertainty will always be part of scientific research, particularly when complex ecological systems are studied observationally and traditional scientific methods are conservative in their rejection of null hypotheses, i.e.  $\alpha = 0.05$  (Kriebel et al. 2001). Still, further consideration by policy makers of how the precautionary principle should be applied in the context of urban riparian management is needed to promote the principles of ESD (see paper 1 of Chapter 1). Indeed, the scale-specific biodiversity planning approach proposed in Paper 1 of Chapter 2 will only succeed if environmental policies necessary to promote it can be upheld in a planning panel, tribunal or court of law in the face of scientific uncertainty and complexity. Possible solutions to this issue are discussed in section 3 of this chapter, but first the specific implications of the research findings from this thesis to biodiversity assessment, riparian management and existing policy frameworks will be discussed.

## 2.2. Implications for biodiversity assessment approaches

In Paper 1, Chapter 2, the importance of using multiple measures of compositional, structural and functional diversity in biodiversity assessment was stressed in order for a holistic picture of biological assets to be attained. Data from Chapters 3 to 5 support this assertion, since different results were attained depending on the different diversity metrics used. For example, in Chapter 3, corridor width was not related to species richness or abundance of ants or plants (a measure of biodiversity composition) nor for Shannon's diversity metrics (univariate measures of biodiversity structure), yet changes were observed in species composition (a multivariate measure of biodiversity structure). Similarly, in Chapter 4, the species richness of ants and plants were influenced by different groups of environmental variables than were found to predict their composition (see Chapter 4, Tables 2 and 3). Likewise, the effect of environmental variables on ants were substantially different between the four measures of diversity: species richness, abundance, Pielou's evenness and Shannon's diversity. This suggests that a single biodiversity metric is not necessarily representative of biological communities as a whole, and highlights the need to use a broad range of biodiversity assessment tools when seeking to apply biological data to the formulation of public policy.

The premise of utilising ants and plants in this thesis was that they would be good indicators of overall biodiversity and ecological function and as such results observed for these taxa would be consistent with patterns in other organisms sharing the same habitats. An assessment and discussion of the surrogacy integrity of ants and plants is provided in Appendix 2. While species richness and composition of ants and plants

from the Ku-ring-gai study sites were correlated significantly, a large degree of scatter was observed in both datasets (see Figures 1 and 2 in Appendix 2). Thus, it is evident that while ant and plant communities are related, it would not be prudent to use patterns in one group to predict those in the other for the purpose of setting environmental policy. In addition, these findings highlight the need for the collection of comprehensive and biodiversity inventory datasets in order to support good environmental decision-making. For example, because a reserve system is designed to protect all known species of native plants does not necessarily mean that invertebrate diversity will likewise be protected (see Chapter 2, Paper 1 for further discussion of the use of biodiversity data in environmental planning).

#### 2.3. Implications of research to riparian management approaches

Results from this thesis highlight the need for a multi-pronged approach to the management of urban riparian zones. Understanding that corridor width has only a relatively minor effect on ant and plant diversity in the context of other factors (see Figure 1) should influence how riparian policy is formulated. Since the biodiversity of riparian corridors is the product of complex sets of environmental and biological parameters and processes, it cannot be predicted by simple models, as is the case for stream temperature for example (Beschta and Weatherred 1984). Therefore it is sufficient to conclude that designing policy to protect riparian biodiversity is a very difficult task. Specifying minimum corridor widths may not be the most effective means of achieving biodiversity outcomes, since biodiversity responds more directly to factors such as disturbance regimes or weed infestation (Figure 1). Although wider

corridors are likely to harbour greater numbers of species as a function of their larger area and greater environmental heterogeneity, corridor width can be considered a rather blunt instrument for the purposes of enhancing species diversity or ecological function within a set unit area such as the immediate streamside zone (see Chapter 4). Instead of this traditional narrow approach to riparian policy, consideration of the following five strategies in concert are likely to yield more effective outcomes:

- 1) Assign minimum riparian corridor widths
- 2) Know the biodiversity assets to be managed
- 3) Actively manage areas of remnant bushland
- 4) Target actions to appropriate spatial scales
- 5) Combat weed invasion.

These five strategies are considered below.

#### 2.3.1. Riparian corridor design

As highlighted in Chapter 3, corridor width does have a positive influence on terrestrial biodiversity at the site scale and the protection of wide corridors is important for the conservation of biodiversity throughout a catchment. These reasons combined with the fact that minimum corridor widths are easily understood by planners and developers mean that it is sensible for corridor width to remain an important component of riparian policy. However, results from this thesis suggest that although wider corridors are better for biodiversity in principle, even narrow corridors can contribute significant biodiversity benefits depending on environmental factors (e.g. soil type, reserve area and connectivity) and management actions (e.g. bush regeneration) (Chapters 3 and 4). Therefore, while environmental managers should

support the retention of riparian corridors that are as wide as possible, they also need to pursue other means of achieving and maintaining biodiversity outcomes.

## 2.3.2 The need to know the state of existing biodiversity assets

Given that the success of a policy is dependent on the degree of scientific knowledge and data that underpins it (McNie 2007), successful management of urban riparian zones will therefore require a thorough inventory of the biodiversity present at a site, particularly considering the idiosyncratic nature of biotic communities observed in this thesis. These data should then be used to set realistic and locally-appropriate policy goals and objectives. For example, if a small population of a once abundant locally indigenous plant species is present at a site, a reasonable goal may be to increase the extent of this species. However, such a goal may not be appropriate at a site that is currently too degraded to support the species. Moreover, monitoring of biodiversity patterns over time will be a powerful resource for adaptive management. This is critical in urban environments where anthropogenic pressure and environmental conditions can change rapidly, requiring prompt management response. However, acquisition of these data will require substantial financial commitment in perpetuity by management authorities.

The inevitable question then arises as to how this monitoring can be achieved with the often-limited resources of local management authorities and without constant access to professional ecological expertise. One possible solution is to use rapid biodiversity assessment methods promoted by many in the field of applied ecology (Gollan et al. 2009; Oliver and Beattie 1996; Triggs 2004), bearing in mind the

caveats and recommendations outlined in section 2.2 above. Programs such as Streamwatch (Sydney Water and Sydney Catchment Authority 2008) that engage schools and other community groups in the collection of water quality data provide an example of how this may be achieved practically. One interesting recent development in this area is the use of citizens to collect data on ant composition to both acquire information on bushland biodiversity and help to engage the wider community in conservation (see Earthwatch 2010). However, in order for this data to make a significant contribution to management outcomes, mechanisms for management authorities to respond are needed. This should involve regular review by an expert with an understanding of the goals and targets of the body responsible for managing the local environment.

In Ku-ring-gai LGA, a rapid biodiversity assessment scheme could complement the existing Rapid Riparian Assessment tool (RRA) (Findlay et al. 2011) such that data is collected for the entire riparian network across the LGA. Terrestrial invertebrates and plants are good candidates for inclusion in a biodiversity assessment scheme, and may be assessed at a higher taxonomic resolution than in the studies conducted in this thesis. Higher taxonomic resolution would permit personnel with lesser developed taxonomic identification skills to acquire useful data (Pik et al. 1999). A scheme such as this would provide important baseline ecological data (currently not included in the RRA), to monitor change in riparian ecological condition over time, help identify areas that are under threat and those that may be the cause of such threats (such as weed plumes). Furthermore, as well as enabling the prioritisation of riparian zones for restoration, such an approach could provide important longitudinal

data on the ecological health of urban bushland, as called for above. However, unlike other biodiversity assessment schemes that reduce species-abundance data down to single metrics (e.g. AUSRIVAS (2008); SIGNAL (Chessman 2001) etc.), data should be retained in raw form to enable a range of analyses to be performed. Adoption of such a biodiversity assessment scheme would be of great value not only to Ku-ringgai Council but any municipality facing the challenge of promoting the ecological integrity of riparian corridors within urban landscapes. Provision of high quality biodiversity data is likely to become more important in the future with expected increases in the population density of urban areas and the development of new land releases.

#### 2.3.3. Management approaches

Conservation of biodiversity in urban riparian systems should not be a passive process but rather an active and adaptive process. The fact that some of the narrow corridors described in chapters 3 and 4 contained communities of plants and ants more akin to those in reference sites suggests that good environmental outcomes are possible across a range of corridor widths. The difference between many of these sites is the degree of weed invasion, which can be improved with appropriate management intervention. The "lock it up and leave it" strategy that is sometimes applied to areas of national park or other large reserves (Primack 2010, Ch. 17) is not appropriate in narrow riparian corridors that experience a range of environmental pressures and edge effects (Florgård 2009; McDonnell 1988). Management authorities must realise that the maintenance of ecologically healthy riparian zones

will require a significant investment of time and resources, regardless of the width of a corridor.

## 2.3.4. Target actions to appropriate spatial scales

A fourth and critical implication of the results from this thesis is the need to consider the different scales at which ecological processes and threats to riparian biodiversity operate. As outlined in Figure 1, riparian ant and plant communities are influenced by factors at the local scale, such as microhabitat characteristics (e.g. the height of lower storey vegetation) or the soil characteristics of a site, and factors operating at the landscape/catchment scale (e.g. the degree of connectivity of a riparian zone to larger tracts of bushland). The breadth of spatial scale that needs to be considered by environmental managers can be a significant challenge, particularly when organisms used to assess environmental condition respond commonly to environmental variation at different scales to those at which land use decisions are made (Andersen 1997). Therefore, a key management recommendation from this thesis is that urban riparian policy approaches that impose sets of guidelines and regulations to be applied equally at all points within a stream network are unlikely to achieve desired ecological outcomes at all sites. Instead, managers should seek to understand and manage urban stream networks and their associated riparian vegetation according to the unique characteristics of each reach (i.e. sections of a stream that are ecologically and geomorphically similar, generally  $10^2$  m to  $10^3$  m in length). Effort is also required to investigate the unique stressors that operate both locally and across the catchment (c.f. Quinn et al. 2001). Some practical implications of considering different scales in riparian management are discussed below:

#### (i) Landscape and catchment scale

The significant differences observed between reference bushland and narrow riparian corridors (Chapter 3) and the impact of landscape connectivity on ant communities (see Table 3 in Chapter 4) suggests that environmental planners should seek to retain large areas of riparian bushland and strive to connect these to other smaller patches where possible. While this approach falls within well-established best practice (Soule 1991), it is encouraging that these principles are supported by the results presented in this thesis (i.e. the relationship between ant composition and connectivity presented in Chapter 4, Table 3).

The influence of adjacent land use (Chapter 4) and differences in biological composition between streamside and upland habitats (Chapter 3) suggest that retention of good quality bushland outside the immediate riparian zone is also important for the conservation of biodiversity at a landscape scale. Management of the urban matrix and protecting habitat on private land will help to enhance biodiversity across landscapes as a whole (Dauber et al. 2003; Forman 1995). The effect of adjacent landscape attributes on native plant species richness (Table 2 in Chapter 4) aligns with the results from Vidra and Shear's study (2008) on urban riparian forests in North Carolina USA. They found that even wide riparian corridors could not ensure protection of riparian forests against weed invasion, as was indeed observed in Chapter 5 of this thesis. Rather, Vidra and Shear (2008) found that the type of adjacent landscape structure was a much better predictor of invasion, with the cover of exotic species in the riparian reserve negatively related to the canopy cover in the surrounding matrix. In this thesis, however, the lack of a significant difference

in ant assemblage composition between urban and rural catchments (Chapter 5) reemphasises that even altering catchment land use dramatically may not necessarily have the expected impacts on riparian ecosystems if other local factors such as upstream propagule sources or human disturbance are not considered within a holistic management plan of approach.

### (ii) Local habitat scale

Chapters 3 and 4 outlined the importance of small-scale abiotic habitat characteristics that can affect ant and plant diversity (see top right box in Figure 1). This is especially so for those environments influenced by edge effects from adjacent urban habitats. Management of edge effects should form a critical component of riparian policy, especially for the enhancement of biodiversity in narrow corridors, which dominate urban stream networks. Possible solutions include ensuring an additional buffer zone exists between the vegetated riparian corridor and adjacent roads or properties (as per the asset protection zones in the Ku-ring-gai Development Control Plan (Ku-ring-gai Council 2005)), fencing remnant bushland, or managing weed infestation. Consideration of edge effects is particularly important if riparian corridors are used for multiple purposes such as biodiversity conservation and social amenity, as is the case in Raleigh, North Carolina USA (see Mason et al. 2007). In their study of forest-breeding birds in Raleigh, Mason et al. (2007) stressed the importance of reducing the impact of trails on forested habitats by minimising trail size and positioning them at the edge of the greenway.

#### 2.3.5. Weed Management

Weed management is already a major focus of many local state and federal government environmental agendas (e.g. the New South Wales *Noxious Weeds Act 1993* (NSW Government 1993)). However, the significant impact of exotic plants on riparian biodiversity that was demonstrated in this thesis suggests that the control of invasive plants ought to be given greater weight within riparian management approaches. Furthermore, riparian ordinances should not focus singularly on corridor dimensions or measures to protect aquatic environments but should also be directed toward strategically addressing weed issues.

Successful control of weeds requires an understanding of the characteristics of the site being restored (Buchanan 1989). The resilience of a site is influenced by a range of historical, ecological and environmental factors, which in turn determine the types of riparian restoration goals that should be set (Palmer et al. 1997). This is especially the case when undertaking ecological restoration works in riparian zones as they possess a range of unique environmental attributes. The first step in the amelioration of weeds in these systems should be to understand the processes that contributed originally to weed infestation and the possible constraints on successful restoration (Richardson et al. 2007). As identified in Chapter 5, the factors that influence riparian weed invasion vary over a broad range of interacting spatial scales and can be biotic or abiotic in nature. The implications of these factors to riparian management were summarised by Richardson et al. (2007), as reproduced below (Table 1).





Some of the concepts referred to in Table 1 are expanded below in practical and applied terms. Given that environmental processes differ with spatial scale, strategies for weed management are considered across different scales as called for in section 2.3.4 above. Scales to be considered are catchment, reach (i.e. a section of a stream possessing similar ecological and geomorphic characteristics, usually >100 m in length) and segment (i.e. a local point on a stream, usually <100 m in length).

(i) Catchment scale: the source of weed propagules and their dispersal pathways need to be considered as weed removal downstream of large weed plumes is unlikely to be successful in the long term (see Säumel and Kowarik 2010). Similarly, removing weeds from bushland upstream of significant conservation reserves and from the urban matrix (see section 2.3.4 above) may help to minimise their spread into high value conservation land. Zones of highest biodiversity value within a landscape (e.g. endangered ecological communities) should also receive greater attention as they function as refugia and can form the core of future riparian recovery (c.f. Brierley and Fryirs 2005; Findlay et al. 2011).

(ii) Reach scale: actively eroding banks should be stabilised prior to removing exotic species. Engineering strategies may be necessary if erosion is severe and leaving alien plants in situ to maintain bank stability can be a beneficial interim measure. Failing to do so may result in bank failure and unsuccessful riparian restoration. Additionally, a site's recovery potential will be influenced by the length of time it has been dominated by weeds, how long propagules of native species remain viable in the soil (Auld et al. 2000) or the preceding levels of biodiversity. For example,

Williams and Roberts (2005) suggest that a high diversity of canopy species enhances the ecological resilience of the understorey.

(iii) Segment scale: Areas of greatest weed dominance (e.g. near stormwater outlets (Lake and Leishman 2004) and near other areas of artificial nutrient enrichment) should not receive primary attention. Rather, restoration activities should focus firstly on areas where greatest recovery is possible and move towards those that are more ecologically degraded. This is in keeping with bush regeneration best practice, whereby areas of bush are worked from high weed infestation to low (Buchanan 1989). In addition, given the influence of vegetation structure on ant communities (Chapter 5), care should be taken not to completely denude large areas of bushland during weed removal as this may have a deleterious effect on ecosystem function and the capacity of a site to recover. Rather a gradual and methodical approach to restoration should be taken.

Since weeds were observed in almost all riparian corridors studied in this thesis, it is not unreasonable to ask if it is prudent to expend valuable resources managing exotic species in urban riparian zones when they appear to be an inevitable feature of urban landscapes. However, this question assumes a narrow, singular goal of restoration work, namely the complete eradication of exotic species. The significant impact of weeds on ant communities in Chapter 5 was due not to the presence of exotic species but rather their physical dominance. Thus, while it may be unfeasible to remove alien plants entirely from urban riparian zones, limiting their ecological dominance is a more achievable goal. Indeed, research presented in this thesis

suggests this approach is very important for invertebrate communities and ecological integrity more broadly. Given the substantial nutrient enrichment experienced by urban riparian zones, a strategy to combat the dominance of weeds may be to replant sites with native but not original vegetation. In sandstone vegetation communities in Sydney, may translate to planting mesic species that have evolved in higher phosphorus and nitrogen environments rather than sclerophyllous species that will be easily outcompeted and require intensive maintenance (Lake and Leishman 2004).

## 2.4. Implications of research to existing environmental policy instruments

The fact that riparian biodiversity is influenced by landscape context in addition to corridor width provides a number of challenges to current environmental and planning policy in Australia, since actions taken some distance away from the riparian zone itself may still affect riparian biota. A possible solution is to adopt Water Sensitive Urban Design (WSUD) strategies in urban catchments. Just as some jurisdictions have introduced planning controls to mitigate the affect of landscape design on receiving waterway health (e.g. the QLD *Sustainable Planning Act* (2009), see Paper 2, Chapter 2 for details), development controls could also be used to minimise the impact of matrix land use on riparian habitat. Many WSUD strategies that aid infiltration/evaporation of rainfall and reduce runoff may assist in promoting the ecological integrity of riparian zones and matrix habitats in addition to the aquatic systems they intend to protect. However, expansion of guidelines in residential Development Control Plans (DCPs) (e.g. Ku-ring-gai DCP 38 (Ku-ring-gai Council

2006)) to require greater proportions of locally occurring indigenous vegetation to be planted in new developments would help maintain native vegetation over time. Furthermore, providing guidelines as to the types of vegetation that may be planted by residents in existing properties may help to transform the biological landscape of cities and enhance biodiversity across large scales.

In terms of current riparian protection mechanisms in NSW, the ecological variability observed between study sites in Chapters 3, 4 and 5 strengthens the case for local councils to be given power to tailor riparian land use zoning categories in Local Environment Plans to their specific local context (see Paper 2 in Chapter 2). In addition, an adaptive and learning culture in management and planning organisations is needed whereby policies are assessed regularly and modified in response to new scientific information (Brunner et al. 2005; Walters 1997). This will ensure best-practice planning approaches are adopted and riparian environments are protected adequately.

## 3. Addressing future challenges

#### 3.1. Research priorities in urban riparian ecology

A number of possible future research agendas arising from this thesis are outlined below, the answers to which would enhance greatly both our understanding the ecology of urban riparian systems and their management.
### 3.1.1. Urban ecological theory

As contended frequently in this thesis, the efficacy of a policy is linked fundamentally to the degree to which the function of the respective system is understood. There remains therefore a pressing need to expand and deepen urban ecological theory in order to establish a solid theoretical foundation from which a range of environmental policies can be formulated (Grimm et al. 2000; Niemela 1999). Many current research themes in urban ecology have implications for riparian management and require further work for their application and uptake in policy. Two of these are discussed below.

Given the importance of adjacent land use on urban riparian communities (Chapter 4), a critical area of current and future research is the nature and extent of edge effects on ecological systems (Lehvåvirta and Kotze 2009). Understanding the nature and magnitude of these impacts is essential when seeking to devise policies that will protect remnant bushland habitats. Research questions that can be asked include "what is the depth of edge influence of a road on adjacent bushland habitats?", "which urban airborne pollutants most greatly affect ecological function?" or "which species of ground flora are most sensitive to trampling alongside walking tracks?". Answers to these questions would further enhance our ability to design urban landscapes in a way that minimises the impact of human activity on important ecosystems.

Another pressing question in urban ecology is how spatial scale affects biological turnover in cities. Understanding scales of beta diversity is an area of intense

research in biodiversity studies today (Tuomisto 2010a, b). These studies are especially critical in the context of urban biodiversity conservation because of the presence of significant rural-urban gradients and high landscape heterogeneity (Luck and Wu 2002). Little is known about the scales at which different organisms change over space, or the extent of biotic homogenisation that occurs in response to urban pressures (McKinney 2006). Knowledge of this will help inform where and how much habitat should be protected in cities in order to achieve holistic biodiversity conservation goals (c.f. Chapter 2, Paper 1). Particularly pertinent to urban rivers and riparian zones is the question of how applicable the river continuum concept is in an urban context (Vannote et al. 1980) given the significant influence of stream order on streamside ant communities (see Table 3 in Chapter 4). Understanding how patterns of biodiversity in aquatic and riparian communities vary longitudinally throughout a catchment and in response to geomorphic change will help inform riparian policy. In particular, this knowledge would assist in selecting which riparian habitats should be prioritised for protection/restoration to maximise biodiversity throughout a catchment.

### 3.1.2. Translation of research to other localities

The degree to which findings from this thesis are applicable to other regions depends on whether riparian corridors respond universally to the same influences identified in Chapters 3 to 5. While many of the principles discussed in this thesis are likely to hold in other regions (e.g. the importance of exotic plants on biodiversity), the specific nature of these are likely to vary. For example, Sydney's urban streams are quite unique in Australian terms (particularly in the Ku-ring-gai LGA) because of geological context. Headwater streams have predominantly flatter gradients than the steeply incised reaches with narrow floodplains further down the catchment (see Chapter 3). Additionally, as studies were performed in a suburban context, an interesting question would be whether the same results would be observed if the studies were repeated in a more densely urbanised context. Drawing comparisons between different cities is a powerful means of uncovering more generalised patterns and processes that will enhance and strengthen urban ecology as a scientific discipline (Niemela et al. 2009). Therefore, applying the riparian biodiversity assessment methods from this thesis in different locations in Australia and globally could assist in the development of a robust framework for urban riparian biodiversity management.

### 3.1.3. Translation of research to other taxa

Assessing the response of organisms other than ants and plants to corridor width and landscape context would be of great interest. There is currently little knowledge of whether mammals, birds, or even fungi respond similarly to corridor width, and what relationships exist between different taxa in this context. Data on this would help to bolster our understanding of the ecology of urban riparian systems and allow more appropriate policies to be developed for the protection of biodiversity in riparian corridors.

### 3.1.4. Species interactions

This thesis focused largely on observing relationships between biological communities and environmental variables. However, as mentioned in section 1 of this discussion, interactions between organisms are key in determining the ecological structure of riparian zones. Studies investigating the effect of one species on another

would help to provide greater depth of insight into the compositional change observed in Chapters 3 to 5. For example, a valuable research pursuit would be to explore whether a threshold in ant community transformation exists. It is not known currently whether an increase in environmental stress such as physical disturbance, pollution or weed infestation is manifest rapidly in ant communities or whether dominant species preclude holistic compositional turnover. Identification of such a threshold would affect the setting of policy guidelines for acceptable levels of human environmental disturbance at landscape and local scales in order to prevent substantial impacts on biological communities (see Figure 1, top boxes). Another question stemming from results presented in Chapter 5 is why such a strong relationship was observed between the dominance of weeds and ant assemblage composition. Manipulative experiments involving habitat change and forced introduction of exotic plants could yield interesting insights into this relationship.

### 3.1.5. Riparian corridors as conduits for species movement

Although the focus of this thesis was on the habitat value of riparian corridors (c.f. Hess and Fischer 2001), an important area of future research is to assess the role riparian corridors play in facilitating the movement of organisms throughout the landscape. Although corridors are often assumed to promote the dispersal of organisms and thus enhance genetic flow (e.g. Wilson and Willis 1975), this has rarely been proven empirically (Rosenberg et al. 1997) (see Haddad, (1999) and Suckling (1984) for examples of successful studies). As very few studies have addressed invertebrate movement, the study of ant movement in riparian corridors using assessment of genetic similarity may help explain more completely patterns of

ant assemblages across landscapes. This would assist environmental managers in the design of riparian reserve networks.

### 3.1.6. Ecological restoration

Given the significance of exotic species to riparian communities that was observed in Chapters 3 to 5, research on the efficacy of bush regeneration practices would be of great benefit to restoration practice and theory. Most assessments of bush regeneration success hinge on the return of native plant species and the reduction of weed cover (Abell 2005; Vidra and Shear 2008). However, there is a dearth of studies examining the recovery of ecosystem function over time or assessing specific regeneration techniques. The ability of terrestrial invertebrates to respond rapidly to weed removal positions them as ideal candidates for such studies (Gollan et al. 2009; Grimbacher and Hughes 2002).

### 3.2. The future of urban riparian management and biodiversity conservation

Urban riparian zones are likely to continue to face significant environmental threats in the future. Continued growth of urban human populations, sprawling cities and ongoing subdivision and redevelopment pressures will lead to an inevitable decline in the ecological health of riparian systems if these stressors are left unchecked. Unregulated development will result in the gradual loss and fragmentation of remnant riparian bushland, degradation of waterways and continued spread of weeds as longterm environmental impacts are pushed aside in favour of short-term economic gain (see Appendix 1). There is undoubtedly a need for local governments to intervene on behalf of the wider community and for generations to come in creating a sustainable and resilient urban environment with ecologically functional riparian zones.

While there are many possible strategies by which this can be achieved, community consultation and scientific advice should be key components of their formulation (Cullen 2006). In thinking about appropriate policies and strategies, it is useful to consider some radical scenarios, for these can often lead to novel approaches to environmental management. Some of these are discussed briefly below.

### 3.2.1. Hypothetical future scenarios

(i) Could riparian corridors of >100 m be protected and restored along all reaches of urban streams, such that all riparian bushland was of the condition of reference sites described in Chapters 3 and 4? This scenario would be possible through promoting much higher density accommodation in less environmentally sensitive areas. However, it would result in the loss of many existing properties and require a radial paradigm shift in the attitudes of landholders and urban planners. In new development areas, this approach would be a more feasible possibility, particularly if corridors were designed with recreation in mind, as per the greenways in the USA (Ahern 1995).

(ii) In contrast, given the substantial extant physical and chemical disturbance experienced by urban riparian zones, would abandoning riparian corridors in favour of other bushland habitats such as those in National Parks be a better use of conservation resources and achieve greater overall biodiversity benefits? While this

would be a noteworthy idea to pursue, the unique species and habitats contained within riparian corridors are unlikely to be represented easily in other reserves. Indeed, biodiversity is a unique entity and cannot be reproduced authentically elsewhere. In this way, biodiversity can be likened to a unique piece of artwork, of which there exists only one original example (see Appendix 1; Collar 2003). Nevertheless, an increased focus on maximum habitat representation in which both riparian and non-riparian communities are protected may be superior to the great weight placed upon riparian corridors to achieve conservation outcomes.

(iii) One mechanism by which the gap between true environmental sustainability and current paradigms of economic growth could be bridged is through adoption of ecosystem service concepts into planning and policy frameworks (see Figure 1 in Appendix 1). Ecosystem services are defined as the "conditions and processes through which natural ecosystems and the species that make them up, sustain and fulfil human life" (Daily 1997 p. 3). For example, clearing of riparian vegetation would become much less attractive if developers were required to pay a fee to compensate for the loss of visual beauty and water filtration services provided by the riparian buffer.

### 3.2.2. Promoting biodiversity stewardship through positive incentives

Perhaps the greatest necessary change to the way urban biodiversity is managed in cities is to move from a regulatory planning system to a system that promotes environmentally considerate behaviour through positive incentives. Positive incentives have been effective in achieving sustainability outcomes in other contexts, such as California's beverage recycling program that provides refunds for used containers (CalRecycle 2011).

As discussed in Paper 1 of Chapter 2, the current planning system in NSW inadvertently perpetuates the destruction of habitat, since merely placing restrictions on where and what type of development can take place encourages developers to clear what vegetation they can before additional restrictions are imposed. Furthermore, when a dispute arises over a refused development application, the onus is often on consent authorities to prove that the development will have negative environmental impacts rather than developers proving that it will not. An example of this is in the recent case *Lipman Properties Pty Ltd v Warringah Council* [2010] NSWLEC 1310, in which an appeal was brought against Warringah Council after its refusal to authorise the development of seniors living apartments on a site containing native vegetation. Despite the presence of an endangered plant species, the appeal was upheld and the development was authorised. Cases such as this highlight the real gap between the current planning system and the principles of ESD, in particular the Precautionary Principle (O'Riordan and Cameron 1994).

In Chapter 2, a series of strategies to promote good stewardship of urban bushland environments in a positive way were proposed. These strategies are especially important for riparian management and may include conservation covenants, reducing landholder rates for good bushland management, and bush tender programs (Department of Sustainability and Environment 2008). One challenging issue associated with economic incentive schemes such as these is the source of

funding. In NSW, local councils currently have no power to levy funds from developers under s 94 of the NSW Environmental Planning & Assessment Act (1979) for the purchase and management of riparian bushland for public environmental benefit or ecological protection. This not only removes an important source of funding, forcing local governments to pursue alternative avenues (see Eco Logical 2009 for discussion of some of these alternatives), but also reflects the ingrained utilitarian view of urban streams and riparian zones that is present in current Australian government structures (see Paper 2 of Chapter 2).

Ultimately however, long term and substantive change to the management of urban environments will be best achieved via a change in social norms and attitudes. Promoting environmental education and interaction with natural habitats is a good step forward. In Raleigh, NC USA, riparian greenways function both as conservation reserves and places for public recreation, with well maintained tracks permitting activities such as walking and cycling (Mason et al. 2007). Although allowing public access in these reserves is not without environmental challenges (Ahern 1995; Jongman and Pungetti 2004), it does help to promote a sense of community ownership and environmental stewardship.

One important way in which the community could be engaged in environmental stewardship of riparian zones in Sydney is through bush regeneration programs. Given the overarching significance of weed invasion on the ecology of riparian zones (Chapters 3, 4 and 5), it is logical that a scheme that both reduces the impact of weeds and helps connect people with their environment should be strongly promoted

by local governments. Many local councils already have large teams of committed bush-care volunteers (e.g. Ku-ring-gai Council and Hornsby Council of northern Sydney) and on the basis of this thesis's findings, this should be encouraged further. Increased advertising, provision of incentives (financial or otherwise) and education are likely to increase community awareness and participation in the restoration of urban bushland. Riparian zones could receive particular attention through encouraging local residents to work on the bushland adjoining their property. Additional financial payments or rate discounts could be offered as incentives. The opportunity for residents to observe the recovery of bushland to which they have a natural connection should help foster an environmental ethic that continues well outside their immediate property.

### **Concluding Remarks**

The management of urban riparian zones is a complex task because of the numerous environmental, social and economic values placed upon them. Although formulating effective policies for riparian protection may be plagued with uncertainty and confusion, empirical research provides the fundamental knowledge basis for sound decision-making by regulatory and management authorities via illumination of biophysical patterns, processes and relationships. This thesis has used a multidisciplinary approach to understand more clearly the efficacy of adopting corridor width as a key parameter in riparian ecosystem protection policy. In doing so it has shed light on broader questions of urban ecological processes, the structure of

ant communities, and patterns of biodiversity in urban riparian systems. Although the relationships uncovered here are ecologically complex, this thesis provides an example of seeking answers to questions that are critical to the practical management of threatened ecological systems, rather than generating neat answers to theoretical questions that may be of little practical significance. This latter approach may be likened to the parable of the man who searched under a street lamp for the keys he lost in the park, simply because the light was better there (Freedman 2010). Indeed, the ecological sustainability of urban riparian and bushland environments requires continued exploration of natural systems and processes, regardless of the challenges that will inevitably be encountered.

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# CHAPTER 7

### Conclusions

This thesis has been positioned at the nexus between environmental policy and relevant scientific research. The combined analysis of ant and plant communities along with a range of relevant landscape variables and policy instruments represents an inter-disciplinary and contemporary approach to the study of urban riparian zones. The completion of this study has contributed to our knowledge of these unique ecological systems within urban environments. Moreover, through the application of scientifically rigorous biodiversity assessment techniques to practical policy and management contexts, this thesis has paved a way for critical future policy-relevant research in the fields of urban and riparian ecology.

The first step in effective biodiversity conservation in urban landscapes is for policymakers and land use planners to grasp the holistic nature of the concept of biodiversity as presented in the scientific literature (Chapter 2). Understanding the compositional, structural and functional attributes of biodiversity within an overarching scientific framework, clarifies yet magnifies the mandate accepted by government authorities to conserve biodiversity under international and national legislative agreements. However, understanding biodiversity within this scientific framework also provides greater clarity and practical guidance as to how biodiversity conservation can be achieved.

Chapter 2 demonstrated that the provision of high quality biodiversity information about local ecosystems is an essential ingredient in the formulation and implementation of effective local environmental policy. However, this information needs to be targeted to the specific needs of policymakers and communicated clearly.

The current inadequate legal protection of urban rivers and riparian zones is the result of definitions and paradigms that do not reflect their physical reality, ecological significance or unique environmental threats (Chapter 2). With growing recognition of the importance of riparian corridors in urban landscapes, data pertaining to their ecological function is necessary for the development of context-appropriate policies.

Analysis of ant and plant assemblages from riparian corridors in the Ku-ring-gai Local Government Area revealed that riparian corridor width (a focus of many riparian protection policies) is significantly related to the biotic composition of these groups, but not species richness (Chapter 3). Moreover, the relationship between width and biotic composition is greatest when organisms existing across the full width of the corridor are considered. This reflects the dual influence of (i) compositional turnover with distance from the stream and (ii) urban edge effects operating on these systems. The compositional differences in ant and plant communities are related to the functional attributes, namely the dominance of opportunistic ants and exotic plants in narrow corridors.

Despite the significant relationship between riparian corridor width and ant composition, consideration of other environmental factors revealed that riparian ant assemblages were more closely related to the density of roads in the adjacent landscape, the type of vegetation community and the connectivity of the reserve to large tracts of bushland (Chapter 4). Ant species richness was also related to variables such as soil type, site gradient, degree of ecological restoration works and the perimeter : area ratio of the riparian reserve. While plant communities were found

to uphold corridor width as the primary variable influencing their composition, this appears to be a function of the inclusion of novel habitats in wide corridors, i.e. those located further upslope from the streambank. Soil pH was found instead to contribute significantly to differences in vegetation within habitats adjacent to streams.

Despite the relationship between road density and ant community structure in Kuring-gai riparian zones, comparison of riparian ant communities from urban and rural catchments in northwest Sydney did not reveal significant differences in diversity or assemblage composition (Chapter 5). Ant community structure in these sites was influenced predominantly by the dominance of exotic vegetation, which itself was driven most greatly by two woody species: *Ligustrum lucidum* and *Ligustrum sinense*.

The consistent influence of exotic vegetation on both ant and plant communities emphasises the influence weed invasion has on riparian ecological function and identifies weed control as a critical factor in urban riparian management. Furthermore, the interaction between ants and plants, as well as the effect of microhabitat and other abiotic characteristics on biological communities, highlights the importance of interspecific and biophysical interactions at small scales in influencing riparian ecological communities. This poses significant challenges for the formulation of environmental policy instruments, as they typically operate at larger scales. The affect of different environmental variables on ants and plants also demonstrates that different taxonomic groups have varying environmental requirements. Care therefore needs be taken when seeking to use one group as a surrogate for another both in scientific study and management applications.

Environmental managers who are responsible for urban landscapes should implement minimum riparian corridor widths as a first step in protecting riparian communities. However, to maximise ecological integrity, corridors should be connected where possible to other tracts of remnant habitat, and the condition of the urban matrix managed so as to minimise the negative impacts of urban development. It is insufficient for environmental managers to simply protect riparian reserves and expect them to achieve good ecological outcomes to be achieved. Instead, active and ongoing management including weed control and ecological restoration is essential because of the constant biological and physio-chemical pressures from upstream environments and the adjoining urban landscape.

# **APPENDIX** 1

Paper: "Legislative and policy challenges for the protection of biodiversity and bushland habitats: An evidence-based approach"
MP Taylor and CD Ives
Published in *Environmental and Planning Law Journal* (2009) 26: 25-48

This paper was birthed from discussions between myself and Prof. Taylor about issues surrounding the effective management of biodiversity within the Sydney region and urban landscapes more broadly. Many of the points discussed came from my research and review of literature in the early period of my PhD candidature, and were developed further in Paper 1 of Chapter 2. This paper discusses the importance of the application of empirical science to the management of urban bushland environments. Of particular interest is the discussion of court judgements regarding disputes between developers and consent authorities over issues of biodiversity conservation. Exploration of these case studies highlights the need for high quality empirical data in decision-making.

### **Author Contributions:**

MP Taylor: 60%Research, preparation of text.CD Ives: 40%Research, preparation of text.

# Pages 223-236 of this thesis have been removed as they contain published material under copyright. Removed contents published as:

Mark Patrick Taylor and Christopher Ives (2009) Legislative and policy challenges for the protection of biodiversity and bushland habitats: An evidence-based approach. *Environmental and Planning Law Journal*, vol. 26, no. 1, pp. 35-48.

# **APPENDIX 2**

# Notes on the Use of Ants and Plants as Biodiversity Surrogates
The degree to which patterns in plant assemblages reflect those in ant assemblages is of substantial significance to both our theoretical understanding of biological interactions and the use of biodiversity surrogates in ecological assessment. Since the rise of biodiversity conservation in environmental policy agendas, the logistical difficulties in assessing biodiversity comprehensively (see Paper 1 of Chapter 2) has increased the appeal of using one group of organisms as a surrogate for others. While this thesis has assessed ant and plant communities separately, the degree to which results can be extrapolated to broader biodiversity patterns and be used to determine biodiversity policy is related to how well ants and plants function as indicators of biodiversity and ecological function. Although only two taxonomic groups were studied in this thesis, the degree of correlation between them will be assessed in the first part of this appendix as a basic indication of their potential usefulness as biodiversity surrogates. The second component of this appendix will assess the applicability of using ants as indicators of ecological function, based upon data derived from this thesis.

## 1. Ants and plants as indicators of biodiversity

A review of the literature on biodiversity surrogacy reveals mixed results (Rodrigues and Brooks 2007). Although some studies have found very strong relationships between different taxonomic groups (e.g. Blair 1999; Saetersdal et al. 2004), most have revealed partial evidence for surrogacy between groups, yet with enough incongruence to advise caution if extrapolating the relationship to policy application (Dormann et al. 2007; Lovell et al. 2007; Oliver et al. 1998; Panzer and Schwartz 1998; Schulze et al. 2004). Even when different taxonomic groups display significant

relationships, they may respond differently to environmental variability, with one group exhibiting higher rates of spatial turnover (beta diversity) than another. This was observed by Oliver et al. (1998) who found that terrestrial invertebrates and plants had much higher levels of turnover than vertebrate species. However, a group exhibiting higher turnover rates (e.g. invertebrates/plants in this case) can still be used legitimately as a biodiversity surrogate in conservation planning, since reserves designed to capture these species will likely also protect the species with smaller rates of spatial turnover (Saetersdal et al. 2004).

Another consistent theme throughout the surrogacy literature is that different measures of biodiversity can influence the strength and nature of the relationship between different groups of organisms. For example, Blair (1999) found that birds and butterflies displayed similar species richness and Shannon diversity responses to an urbanisation gradient, but did not respond similarly in terms of abundance. Likewise, despite high correlations between the species richness of plants, birds and insects, Schulze *et al.* (2004) could only recommend a small number of taxa that were good indicators of assemblage patterns of others. Furthermore, Panzer & Schwartz (1998) found that only a third of the measures of plant diversity were effective in explaining patterns in insect diversity. These results are important to bear in mind when interpreting patterns in biotic data from this thesis.

The sampling design used in the Ku-ring-gai study (Chapters 3 and 4) enabled finescale assessment of surrogacy between ants and plants because species abundance data was collected at the same location and spatial resolution for the two

groups. Results from comparison of species richness and community composition of both ants and plants within the 18 riparian study sites in the Ku-ring-gai Local Government Area are presented below. Rather than attaining a rarefied site average of richness and composition as was done in Chapters 3, 4 and 5, data were compared at the transect scale (this involved pooling ant specimens from sets of three pitfalls corresponding with a vegetation transect) in order to conserve the fine scale data patterns. Comparison of assemblage composition was enabled via Mantel analysis, which assesses the correlation between pairwise comparisons of samples of one taxonomic group with those of the other.

Significant relationships were observed between ant and plant data for both species richness and community composition (Figures 1 and 2). Interestingly, the slope of the relationship is similar between the two diversity measures, as are the (relatively low)  $R^2$  values. As per Dormann *et al.* (2007), these results identify a noteworthy association, but not a strong enough relationship to *predict* the biodiversity of one group based on the other. The association between ants and plants is most likely related to the shading of habitats by vegetation (especially weed species) and the food and other resources provided to ants by vegetation. These factors are discussed in more detail in Chapters 3 and 4.

Appendix 2



**Figure 1** Scatterplot of the relationship between the transect species richness of plants and ants in Ku-ring-gai riparian study sites.



**Figure 2** Scatterplot of the pairwise Bray-Curtis similarity of vegetation transects (xaxis) plotted against the respective similarity in ant composition between each pair. Spearman correlation analysis based on Mantel coefficient revealed a statistically significant positive relationship (Rho = 0.351, P = 0.001), as represented visually in the scatterplot.

In Figures 1 and 2, analyses are framed with plants as the predictive variable and ants as the response variable. This aligns with studies such as Greenslade and Greenslade (1977) and Hoffmann et al. (2000) who found that ant communities are influenced by vegetation structure and species richness. However, it is possible that plants may respond more strongly to changes in ant species composition, as is outlined in Chapter 6. Nevertheless, if one group was to function as "indicator" taxa, it is more likely to be plants for practical reasons. For example, plants are conspicuous, sedentary and botanical identification skills are more widely held amidst professionals and those local government agencies than skills in insect taxonomy (see Hopkins and Freckleton 2002). However, advances in rapid biodiversity assessment using terrestrial macro invertebrates may make the use of ants in environmental monitoring more feasible than has been the case in the past (Oliver and Beattie 1996). In terms of application to riparian management, the observed relationship between ants and plants (Figs 1 and 2) provides scientific support for efforts to restore riparian habitats and promote vegetation diversity, as this is likely to have concomitant positive affects on ant communities. However, the degree of scatter in Figures 1 and 2 suggests that basing conservation planning decisions on vegetation patterns (i.e. which patches of bushland should be protect within a landscape) may not adequately protect invertebrate diversity.

## 2. The use of ants to indicate ecological function

Another form of biological surrogacy that featured strongly in this thesis was the use of the ant functional groups model (Andersen 1995) to test for differences in ant function based upon the response of ant genera or species groups to environmental

stress. Functional groups have been used successfully to indicate environmental stress and community recovery after mining, forest plantation, logging and fire (Andersen and Majer 2004; Stephens and Wagner 2006). In Chapter 3, functional groups successfully identified the groups of ants that drove differences in ant composition between sites of differing corridor width. However, in Chapter 5, functional group responses did not fit predictions. There, greater abundances of species belonging to "Opportunist" and "Generalised Myrmicinae" groups were identified in sites that had been predicted to experience greater levels of environmental stress (i.e. urban riparian zones).

The failure to observe consistent functional group patterns in the analysis of the relationships between ant species and the proportion of exotic vegetation at sites in north west Sydney may point towards the importance of unique site characteristics in affecting species function and interspecific interactions. Since lower temperatures and high levels of shade (typical in riparian zones) can increase the level of stress perceived by many ant species (King et al. 1998; Lach et al. 2010) riparian conditions may reduce interspecific competition and thus diminish the ecological signatures expressed by various ant functional groups.

While functional groups have been used successfully to indicate local environmental disturbance in Sydney (Lassau and Hochuli 2004), this is outside the scope of application originally intended by Andersen (1995), which was to indicate broad structural community patterns at biogeographical scales (Philpott et al. 2010). It is understandable therefore that other studies have found the use of ant functional

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groups based on Andersen's (1995) classification to be inadequate in detecting environmental disturbance, when high degrees of habitat heterogeneity or periodic disturbance exist (Gollan et al. 2009; New 2000). Indeed, the reason Lassau & Hochuli (2004) found functional groups to be effective at small scales may be because changes in habitat structure were the focus of their study. Andersen's functional group model may not therefore detect adequately subtle disturbances in the form of pollution or change in vegetation composition.

In light of these limitations, there have been calls to review the use of functional groups in small-scale environmental assessments. Recent research has indicated that use of morphological traits such as body size, ratios of leg to body length or mandible features may be more effective in indicating ant function at local scales than grouping ants simply according to genus (Nipperess and Beattie 2004; Silva and Brandao 2010; Weiser and Kaspari 2006). Weiser & Kaspari (2006) provide an indication of the future direction of these "morphological-based" classifications. In their measurement of 10 morphological traits of ants in 122 genera in the New World, they were able to distil a large portion of the variability in morphological traits down to 3 features: head size, eye size and appendage length. These features correlated well with functional attributes such as diet and foraging substrate, indicating that morphological traits may successfully be used as indicators of ecological function. More recently, Silva and Brandao (2010) successfully grouped ants into guilds for application in fine-scale functional assessment & description of ant communities. Interestingly, Silva and Brandao (2010) found that guilds were partitioned according to microhabitat distribution, body size and shape, eye size and shape, and

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phylogeny. Again this confirms the importance of habitat structure in arranging ant communities at local scales. Using morphological features may also prove to be a useful approach in ecological assessments undertaken by management organisations, as speciality knowledge of ant taxonomy is not necessarily required.

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# **APPENDIX 3**

## **Attached Electronic Data Files**

Biological and environmental datasets used in the preparation of this thesis are attached electronically as a CD bound to the back cover. Click first on the file "Read\_This\_First.doc" for instructions on how to access files and for additional information pertaining to the data.