Characterising Connectivity for Fragmented Landscapes in Support of Regional Planning and Impact Assessment

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November 2022

A thesis submitted to the University of Nottingham Malaysia

for the degree of Master of Philosophy

Statement of Word Count

Number of words contained in this document including references and appendices: 24,977 words

Abstract

Biodiversity loss resulting from habitat fragmentation and land use change is occurring at an increasing rate globally. Fragmentation can lead to genetic isolation within or between populations, due to loss in connectivity. The issue of ensuring ecological protection of fragmented areas must be addressed. Methods for accurately characterising connectivity within a landscape and integrating the study of connectivity loss into impact assessments is crucial. This thesis addresses this gap. It: (1) investigates the extent of ecological connectivity has been considered as part of infrastructure impact assessments (2) characterises connectivity for landscapes fragmented by agriculture, by considering how fine-scaled vegetation such as scattered trees support connectivity and the implications of ignoring such elements have on land use planning and (3) applies a spatially explicit scenario analysis of alternative road alignments and mitigation options to address common criticisms of the lack of consideration of connectivity in environmental impact assessments.

This work begins by providing a systematic review on the extent of ecological connectivity research in the context of linear infrastructure Environmental Impact Assessments (EIAs). that the review revealed that there is a lack of consideration of specific locational impacts or design alternatives for reducing harmful impacts to biodiversity, in linear infrastructure EIA. There is also much uncertainty regarding best methods and metrics to quantify and compare between multiple design options, and it is evident that research seldom recommends mitigation measures or their specific locations in environmental impact assessments. It is necessary to develop quantitative approaches for assessing the effects of transport networks on landscape connectivity at large spatial scales. It is shown in this chapter that while such approaches do exist, there are few published instances compared to the global geographic scope and scale of infrastructure projects; consequently, there are few examples from which to draw upon. Overall, the review showed a lack of standardised

and efficient procedures to guide EIAs and decision making. The need for such procedures is made imperative by the increases in the number and extent of infrastructure developments, many of which require environmental impact assessments.

Subsequently, an assessment of connectivity within a fragmented landscape dominated by agriculture and pasture was conducted. It demonstrated how fine-scaled vegetation such as scattered trees support connectivity and the implications of ignoring such elements in regional scale land use planning. Modern connectivity modelling techniques rarely consider fine-scale movement patterns associated with movement between fine-scaled structural connectivity features, such as scattered trees, roadside corridors, and small habitat patches. This connectivity assessment mapped scattered trees in an agricultural area where pasture was the predominant human-altered land cover, then used a least-cost path analysis and a graph-theoretic approach to show that by ignoring scattered trees, simulated movement patterns do not match typical movement patterns seen in field research. The work showed that connectivity models that omit fine-scale landscape elements may misrepresent connectivity patterns.

Building on our findings above, we attempted to address common shortcomings of infrastructure EIAs. Construction of roads is one of the leading causes of habitat fragmentation and biodiversity loss on a global scale. This study represents the first research to combine both a scenario assessment and an evaluation of mitigation strategies for a road infrastructure project. It applied a spatially explicit connectivity model that considers fine-scale movement patterns, along with scenario analysis of alternative road alignments for a bypass in the Australian town of Beaufort. The wildlife connectivity model used expert-based parameterization of species movement traits and a combination of least-cost pathways, circuit theory, and graph theory to represent five conservation targets with contrasting dispersal abilities and habitat needs. For each of target, impacts of four distinct road alignments were modelled, with mitigation measures and alternatives routes then evaluated to identify the least damaging. The results demonstrated that each conservation target was affected differently, with terrestrial species with greater dispersal distances being the most affected. However, the modelling indicated that one alignment option had least impact overall, and that combining this route with wildlife crossing structures increases connectivity for all conservation targets. This real-world case study demonstrated the feasibility of integrating ecological connectivity modelling with scenario analysis in EIAs using a clear and quantitative manner.

The work presented in this thesis shows how fine-scale movement patterns can be characterised and modelled, and incorporated into EIAs and mitigation proposals for infrastructure development. It shows how specific measures can avoid, minimise, or mitigate adverse effects from infrastructure development, and more broadly to help land managers identify important conservation values that are often ignored. The quantitative assessments and models show how knowledge about dispersal networks, and the availability and distribution of suitable and accessible habitat can be used assess the effects of land use changes on a region's fauna and flora. Different species experience habitat fragmentation in different ways and at different scales, underlining the necessity for species-specific EIAs undertaken at appropriate at spatial scales.

Preface

The work presented herein was completed as a body of work for this thesis and is substantially my own work. Publications and contributions from others based on the content of this thesis is as follows:

The content in **Chapter 3** was published as:

Tiang D.C.F., Morris A., Bell M., Gibbins C.N., Azhar B., Lechner A.M. 2021. Ecological connectivity in fragmented agricultural landscapes and the importance of scattered trees and small patches. *Ecological Process*. 10(1). doi:10.1186/s13717-021-00284-7.

The content in **Chapter 4** was submitted for publication as:

Tiang D.C.F., van der Ree R., McCaffrey, N., Gibbins C., Lechner A. (in review) Ecological connectivity in Environmental Impact Assessments: Modelling alternative highway bypass scenarios, *Impact Assessment and Project Appraisal.*

Acknowledgements

Words cannot express my gratitude to my supervisors, Alex Lechner and Chris Gibbins for their invaluable patience and feedback. Their generously provided knowledge and expertise hugely contributed to the completion of this work and my growth as a researcher. Additionally, this endeavour would not have been possible without the generous support from the Landscape Ecology and Conservation Laboratory and the Faculty of Science and Engineering, University of Nottingham, who financed my research.

I would also like extend my sincere thanks to members of MidCoast Council- Andrew Morris and Matthew Bell, and WSP Australia- Rodney Van Der Ree and Nic McCaffrey. It has been a pleasure collaborating with you.

I am also grateful to my cohort members, especially my office mates of the Landscape Ecology and Conservation laboratory, for their feedback and moral support. Thank you, Michelle, Aliff, Li Shuen, Zoe, and Karen.

Lastly, I would be remiss in not mentioning my parents, David and Laura, and my friends. Their belief in me has kept my spirits and motivation high during this process.

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Chapter 1. Introduction

Conversion of natural ecosystems to human land use fragments and decreases available habitat for wildlife, restricting species movement across the landscape (Lumsden and Bennett 2005; Lindsay et al. 2008; Hadley and Betts 2009; Beier et al. 2011; Rogan and Lacher Jr. 2018; Liu et al. 2020; Bolliger and Silbernagel 2020). Reduced habitat connectivity can have detrimental consequences for population viability, resulting in a higher extinction risk than habitat loss alone (Fischer and Lindenmayer 2006; Brook et al. 2008). Improving the management of human-altered landscapes is essential for mitigating the effect of human induced fragmentation on species mobility and connectivity and, ultimately, for guaranteeing the survival of populations and ecosystems.

Many conservation and management plans concentrate on connecting remnant patches of habitat fragmented by human activities. It has become commonplace for land use planners to determine the most suitable areas to carry out conservation efforts either by restoring connectivity or identifying important wildlife corridors that need to be protected (Saura and Rubio 2010; Galpern et al. 2011; Pereira et al. 2011; Dalang and Hersperger 2012; McRae et al. 2012; Foltête et al. 2014). This can be done by characterising connectivity within human induced fragmented landscapes, which identifies vegetation patches or other structural connectivity factors that influence connectivity patterns and contribute to connectivity at the regional scale. Spatial patterns of areas isolated by fragmentation are useful for pinpointing barriers to connectivity (Lechner et al. 2015c), and such areas can be prioritised in conservation planning in order to assure future connectivity.

Infrastructure development continues to expand globally, sustaining growing human populations and underpinning economic expansion (Henderson 2003; Perz et al. 2007). The expansion of linear infrastructure (notably roads) has significant implications for biodiversity, and is one of the leading causes of habitat fragmentation and biodiversity loss globally (Laurance et al. 2015; van der Ree et al. 2015; Ng et al. 2020). Roads directly contribute to the reduced mobility of individuals and genes within and between populations via fragmentation, (Ceia-Hasse et al. 2018). Due to limited access to resources and mates, whole populations may become unviable. This effect can cascade across biological communities, altering patterns of species composition and richness, and eventually influencing ecosystem function (Fahrig and Rytwinski 2009; Barrientos et al. 2021).

A key stage in addressing the ecological implications of infrastructure development is to predict and quantify potential impacts during the planning phase (Mortberg et al. 2007). For infrastructure and other large project developments, a formal environmental impact assessment (EIA) is usually applied. Legal frameworks worldwide have emphasised the necessity to incorporate biodiversity within the EIA planning and decision-making process (Karlson et al. 2014) and in recent years, ecological connectivity has been an emerging topic of interest within transport infrastructure EIAs (Jepson 2016). Nevertheless, to date EIAs have been conducted with different degrees of effectiveness (Wathern 2013), and continue to lack quantitative projections of landscape-scale ecological connectivity (Deslauriers et al. 2018; Spanowicz and Jaeger 2019). Typically, EIAs focus solely on site-level impacts within the construction footprint, (Halpern et al. 2008; Halpern and Fujita 2013; Andersen et al. 2015) whereas cumulative impacts on biodiversity can occur across the wider landscape (Hawke 2009). In addition, the necessity for impact mitigation measures, such as animal crossing bridges, is frequently not considered beyond the scale of the development site.

Despite gaps in knowledge and shortcomings in evaluations the impacts of infrastructure on ecological connectivity and mitigating them (Folkeson et al. 2013; Mimet et al. 2016), land use planners are now supported by new technologies and methods for identifying said impacts (González Del Campo and Gazzola, 2020). With advances in ecological models and Geographic Information Systems (GIS), there is great potential for incorporating quantitative and spatially explicit ecological assessments into regional planning and impact assessments to quantify impact scenarios and evaluate alternative development options. Existing connectivity modelling methods, such as least-cost path analysis, circuit theory, and graph theory, provide a modern, diverse toolbox for studying and quantifying different aspects of landscape connectivity (Merrick and Koprowski 2017; Zeller et al. 2018; Balbi et al.

2021; Foltête et al. 2021) . Scenario-based evaluations are particularly relevant to the impact assessment process because they can develop recommendations on how alternative infrastructure alignments impact ecological connectivity, but they are infrequently used (but see Tannier et al. 2016; Huang et al. 2018; Tarabon et al. 2020; Fullman et al. 2021; Sahraoui et al. 2021). In addition, scenario-based analysis of wildlife mitigation measures is essential for evaluating the efficacy of various alternatives (Gurrutxaga and Saura 2014; Clauzel 2017). Modelling and GIS approaches for assessing the consequences of alternative road alignment designs and mitigation measures have the potential to provide improved insights into the long-term viability of proposed developments and support a more robust EIA, particularly when road alignments are selected to minimise their impact on ecological connectivity.

There are notable studies that have used these approaches to model landscape connectivity and thus made significant contributions to what is now a large body of research and knowledge (Merrick and Koprowski 2017; Zeller et al. 2018; Balbi et al. 2021; Foltête et al. 2021). However, such studies do not effectively incorporate truly fine-scale elements such as scattered trees (including paddock trees) and roadside plant patches, nor do they address how these elements promote migration between other larger habitat patches (Lechner et al. 2015b; Synes et al. 2016; Drielsma et al. 2022). In fragmented pasture landscapes used for livestock grazing, dispersed trees, small patches of natural vegetation, and corridors are recognised as key movement facilitators (Fischer and Lindenmayer 2002; van der Ree et al. 2004) because they allow faunal species to traverse great distances from one patch to another by functioning as stepping stones. Scattered trees are dispersed, solitary trees, typically remnants of an unbroken forest, that are surrounded by open land (Manning et al. 2006). They are prevalent in human-dominated and fragmented forest settings and can be the result of forest clearance and thinning (Manning et al. 2009). They are regarded as "keystone structures" in human-dominated landscapes due to their usefulness: they provide foraging sites and shelter for numerous species (Carruthers et al. 2004; Le Roux et al. 2018; Barth et al. 2020), habitat for insects and pollinators (Lumsden and Bennett 2005; Prevedello et al. 2018), and focal points for tree regeneration (Dorrough and Moxham 2005; Derroire et al; Manning et al. 2006). In addition, they serve as a stopover and a place to rest, protecting many woodland and forest species from predators when they go out into the open matrix, so rendering fragmented landscapes "useful."

The research presented in this thesis addresses the knowledge gap in existing ecological connectivity research pertaining to the use of connectivity modelling in EIAs and, consequently, its role in regional land use planning. This thesis applies spatially explicit models to (i) highlight how dispersal is facilitated by scattered trees and small patches of vegetation, and so contributes to broader (regional-scale) connectivity, (ii) quantify environmental impacts in linear infrastructure beyond the construction footprint, (iii) quantify and visualise impact scenarios, and (iv) evaluate alternative alignments and mitigation options. The thesis provides a transparent, rigorous and systematic approach to the assessment of connectivity across fragmented landscapes, designed to support regional planning and infrastructure impact assessments. The work also highlights the importance of modelling the role of wildlife crossing structures for connecting habitat and reducing the barrier effect of linear infrastructure on terrestrial animal groups (Iuell 2003; Olsson et al. 2008; Glista et al. 2009).

1.1 Research Aim and Objectives

The aim of this thesis is:

To integrate multiple, spatially explicit modelling techniques to improve understanding and assessment of habitat connectivity.

The objectives of this thesis are:

(1) To investigate the extent to which ecological connectivity has been considered as part of infrastructure impact assessments.
(2) To characterise regional connectivity for landscapes fragmented by agriculture, by considering how fine-scaled vegetation such as scattered trees support connectivity and the implications of ignoring such elements on land use planning.

(3) To apply a spatially explicit scenario analysis of alternative road alignments and mitigation options to address common criticisms of ecological assessments in infrastructure impact assessments, by using a real-world linear infrastructure development as a case study.

1.2 Research Questions:

(1) To what extent has academic literature considered ecological connectivity in the context of infrastructure impact assessments?
(2) How do fine-scale connectivity elements influence connectivity patterns within fragmented landscapes, and support connectivity on a regional scale?
(3) How can spatially explicit ecological models be utilised to support and improve infrastructure impact assessments?

1.3 Thesis Outline

This thesis consists of five chapters. The main empirical chapters (Chapters 3 and 4) are written so that they can be read independently as standalone research articles. These two chapters have either been accepted by, or submitted to, a peer-reviewed journals (see details in title pages of respective chapters). The contents of the chapters remain the same as the published versions except for the formatting, which has been revised to maintain a consistent style throughout this thesis; the references cited have been compiled into a single list at the end of the thesis.

Chapter 2 provides a systematic review of ecological connectivity modelling research within infrastructure impact assessments. As the extent and practice of ecological connectivity research in the context of assessing infrastructure impacts have not been previously investigated, this chapter helps stress the novelty and importance of the research presented in this thesis. The review underlines challenges of ecological assessments within formal impact assessments and guides the subsequent chapters by identifying current gaps in knowledge as well as the range of existing methods and techniques applied by others within the same context. The review thus acts as a compass, outlining the next steps for ecological connectivity research in assessing infrastructure impacts.

Chapters 3 and 4 investigates how characterising the overall connectivity of an area can inform better management of human modified landscapes and reduce the negative effects of fragmentation and habitat loss. Chapter 3 characterises connectivity for a general representative species, while chapter 4 characterises connectivity for five conservation target species across five scenarios involving an infrastructure development.

Chapter 3 investigates the consequence of not incorporating fine-scale movement patterns when modelling connectivity. A least-cost path analysis and a graph-theoretic approach was used to characterise connectivity, using a gap crossing threshold within a typical woodland ecosystem that is fragmented by agriculture. The gap crossing threshold describes how fine-scaled vegetation such as scattered trees support connectivity. This chapter illustrates that connectivity models that exclude fine-scale landscape features risk misrepresenting connectivity patterns, which will likely misinform land use planning and misdirect land management efforts.

Chapter 4 builds upon the previous chapter's findings to addresses common challenges of ecological assessments in infrastructure EIAs. A real-world linear infrastructure development is used as a case study, applying a spatially explicit connectivity model with scenario analysis of alternative road alignments for a highway bypass. It represents the first work to include both a scenario assessment and evaluation of mitigation options for a road infrastructure development. The research was able to quantify impacts for a range of conservation target species while identifying a single alignment option with the least overall impact on connectivity. This case-study demonstrated the potential to apply a transparent and quantitative approach to mainstream ecological connectivity modelling within scenario analyses in impact assessments.

Finally, chapter 6 summarises the findings of this thesis and describes potential future directions for the application of ecological models in impact assessments across the world, discussing how these assessments can be supported by off-the-shelf ecological connectivity tools.

Chapter 2. A Review of Ecological Connectivity Within Infrastructure Impact Assessments

We conducted a systematic literature review to investigate how extensively ecological connectivity is being used to assess infrastructure impacts and the specific approaches that have been applied to frame our research methods and demonstrate the novelty of our study. The search string applied on SCOPUS was:

TITLE-ABS-KEY (‘infrastructure project’ OR ‘linear infrastructure’ OR ‘road*’ OR ‘highway OR ‘rail’ OR ‘railway’ OR ‘high speed rail’ AND ‘ecological model*’ OR ‘connectivity model*’ OR ‘mapping potential impact’ OR ‘connectivity’ OR ‘landscape analysis’ OR ‘ecological network’ AND ‘impact assessment’ OR ‘spatial planning’)

Our initial search returned 323 results. However, we only kept studies that explicitly set out to model impacts of a proposed or existing linear infrastructure and this decision was based on reading each paper’s title, abstract and keywords. Those that assessed general land use development such as urban expansion over time but included transport networks were also selected. Of all the search results 29 (9%) were relevant to our review (Table A 1).

Studies that assessed connectivity with the purpose of predicting the impacts of linear infrastructure projects were few. Of the 29 relevant studies, more than half (58.6%, n = 17) were pre-impact studies that predicted likely effects of a proposed infrastructure development while the remaining ones looked at an existing or both existing and proposed infrastructure (34%, n=10). Predictive studies that include assessment of alternative designs or routes can guide the design of mitigation measures (Clauzel 2017).

Road developments were the most common (51.7%, n=15) linear infrastructure project assessed, followed by rail (20.7%, n=6), transport network extensions (both road and rail) as part of urban expansions (10.3%, n=3), power lines (6.9%, n=2) and unspecified linear infrastructure studies (3.4%, n=1). The studies were also conducted across a broad range of spatial scales, ranging from the immediate surroundings of the construction footprint of a project (local scale) (Karlson et al. 2016) to the entire home range of an impacted species (landscape or regional scale) (Fullman et al. 2021).

The conservation targets assess in these projects varied, with the majority of studies assessing the impacts of infrastructure developments on multiple species or species 'guilds' (n= 12, 41.4%), followed by papers assessing impacts for a single species or a single 'general representative species' (see Tiang et al. 2021) (n=11, 37.9%), while a few assessed impacts without a conservation target species (n=6, 20.7%). The multi-species approach predominated published work, as anthropogenic impacts affect a wide range of species and taxonomic groups (Boitani et al. 2007; Crooks and Sanjayan 2010; Lechner et al. 2017). Transport infrastructures disproportionately affect certain taxonomic groups, as a function of body size, abundance and diet (Ford and Fahrig 2007; Barthelmess and Brooks 2010; Holderegger and Di Giulio 2010; Rytwinski and Fahrig 2012). Thus, there is a need to assess impacts on connectivity for a larger set of species over large geographic scales.

The most common method used to empirically assess linear infrastructure impacts was ecological connectivity models based on a graph theoretic and least-cost path analysis (58.6%, n=17), followed by assessing movement probability (13.8%, n=4). Other methods included Circuit Theory, modelling habitat availability, ranking patch quality using patch indices and landscape metrics, multi-criteria analyses, individual based modelling, and an integrated landscape ecology approach. Graph theory and least-cost path analysis provide a compromise between the ability to represent ecological flows at large spatial scales and the amount of input data required (Calabrese and Fagan 2004), while also being able to estimate local impacts by measuring variations in local connectivity metrics at the scale of habitat patches and visualising changes in the networks' structure (Tannier et al. 2016).

More than half of the papers (55.2%, n=16) incorporated scenario analyses or considered design alternatives in their assessment. This allowed them to evaluate and rank alternative linear infrastructure corridors from an ecological perspective and propose less impactful alternatives. Some of the authors highlighted the lack of consideration of specific locational or design alternatives in EIAs (e.g., Dalloz et al. 2017) in reducing harmful impacts to biodiversity, in part due to uncertainty regarding best methods and metrics to quantify and compare between multiple design options. Only seven papers recommended mitigation measures or locations and six of these used least-cost path modelling and a graph theoretic approach to assess connectivity (Nielsen et al. 2012). Mitigation models can help locate areas in which to promote habitat restoration or establish wildlife crossing structures as compensatory measures (Found and Boyce 2011). Landscape graphs allow planners to do this by iteratively testing each linkage to determine which one maximizes overall connectivity (e.g., Clauzel 2017) or by ranking each linkage's contribution to movement probability (e.g., Gurrutxaga and Saura 2014).

From our review of impact assessment methods of transport infrastructure, standardised and efficient procedures to guide ecological impact assessment and decision making about appropriate mitigation measures appear to be lacking. Quantitative methods to assess the impacts of transport networks on landscape connectivity at broad spatial scales have to be established. Our review demonstrated that while such methods exist, published examples are few when set against the geographic spread and scale of infrastructure projects globally; thus, examples to draw upon are few. The need for such examples is made pressing by the growing pace of infrastructure development, many of which require EIAs.

Chapter 3. Ecological connectivity in fragmented agricultural landscapes and the importance

Published as: Tiang DCF, Morris A, Bell M, Gibbins CN, Azhar B, Lechner AM. 2021. Ecological connectivity in fragmented agricultural landscapes and the importance of scattered trees and small patches. Ecological Process. 10(1). doi:10.1186/s13717-021-00284-7.

3.1 Introduction

Conversion of natural ecosystems for human land uses can lead to fragmentation, loss of habitats and restriction of species movement (Lumsden and Bennett 2005; Lindsay et al. 2008; Hadley and Betts 2009; Beier et al. 2011; Rogan and Lacher Jr. 2018; Liu et al. 2020; Bolliger and Silbernagel 2020). The decrease in habitat connectivity has adverse effects on population viability, resulting in greater extinction risk than from the loss of habitat alone (Caughley 1994; Fischer and Lindenmayer 2006; Brook et al. 2008). Better management of human-modified landscapes is central to minimising the impact of fragmentation on species movement and connectivity and ultimately, ensuring the viability of populations and ecosystems.

Contemporary methods for modelling landscape connectivity include least-cost path analysis and graph theory; these provide a diverse toolbox for studying the different aspects of connectivity (Urban and Keitt 2001; Adriaensen et al. 2003; McRae et al. 2008; Foltête et al. 2012). Least-cost path analysis characterises non-habitat/matrix based on dispersal costs, which represent the metabolic price and mortality risk of moving across such areas (Adriaensen et al. 2003; Sawyer et al. 2011). Dispersal cost is influenced by a combination of land cover attributes, such as urbanisation, and species-specific dispersal probabilities over various distances. Cost-weighted analysis is used to produce the least-cost pathways connecting suitable habitat patches. Subsequently, using a graph theoretic approach, network measures are calculated to quantitatively assess the significance of patches within a connectivity network (Minor and Urban 2008; Rayfield et al. 2011).

Many studies have incorporated these approaches in modelling landscape connectivity and have made significant contributions to a now large body of research (Urban and Keitt 2001; Adriaensen et al. 2003; McRae et al. 2008; Urban et al. 2009; Rayfield et al. 2011; Foltête et al. 2012). However, these approaches have a number of important limitations; most notably, they do not adequately incorporate truly fine-scale features such as scattered trees (including paddock trees) and road-side vegetation patches, nor address how these facilitate movement between larger habitat patches (Lechner et al. 2015b). In landscapes fragmentated by pasture, scattered trees, small patches and corridors are recognised as important facilitators of movement (Fischer and Lindenmayer 2002; van der Ree et al. 2004) as they allow for species to move long distances from one patch to another by acting as stepping stones.

Scattered trees are dispersed individual trees, often remnants of intact forest, that are surrounded by open space (Manning et al. 2006). They are common in human dominated and fragmented forest landscapes and can result from practices such as clearance and thinning of forests (Manning et al. 2009). They are recognized for their usefulness and are widely regarded as “keystone structures” in human-dominated landscapes: they provide foraging sites and shelter for many species (Carruthers et al. 2004; Le Roux et al. 2018; Barth et al. 2020), habitat for insectivores and pollinators (Lumsden and Bennett 2005; Prevedello et al. 2018), focal points for tree regeneration (Dorrough and Moxham 2005; Derroire et al. 2016), soil nutrient retention (Wilson 2008) and connectivity for a wide range of biota (Manning et al. 2006). In addition, they act as a stopover and a place to rest, providing protection from predation for many woodland and forest species on their ventures out into the open matrix, effectively making fragmented landscapes “usable”.

The aim of this study was to characterise connectivity in fragmented agricultural landscapes dominated by open pastures, where we hypothesised that dispersal is facilitated by scattered trees and small patches of vegetation such as road-side corridors. We assessed (1) how scattered trees and other fine-scaled structural connectivity elements influence connectivity patterns within fragmented landscapes and (2) the contribution of such landscapes to regional-scale connectivity beyond their boundaries. To address the study aim, we used information on fine-scale dispersal behaviours to model the contribution of scattered trees and small patches for the Karuah-Myall catchments, New South Wales, Australia, for a "general representative species" dependent on native woody vegetation (Lechner et al. 2015b). The Karuah-Myall catchments represent a typical woodland ecosystem on the east coast of Australia which has been fragmented by pasture agriculture. We mapped and modelled connectivity using an interpatch dispersal distance, gap crossing threshold and resistance from different land cover types. The gap crossing distance threshold was used to model movement between fine-scaled vegetation features. Movement was characterised with least-cost paths, and the importance of links and patches to connectivity was quantified using a graph theoretic approach (Foltête et al. 2012). We compared the least-cost paths modelled with and without scattered trees. In addition, the importance of protected areas for conserving connectivity and the contribution of the study area to connectivity beyond its boundaries, specifically the Great Eastern Ranges national wildlife corridor scheme was also assessed. We conclude by discussing the important role of scattered trees and small patches in connecting landscapes and the importance of explicitly modelling them.

3.2 Methods

3.2.1 Study area

The Karuah River catchment is situated in the lower north coast of New South Wales, Australia. It borders the Hunter River catchment in the south, and the Manning River catchment in the north. The catchment is approximately 2410 km2. Land uses within the Karuah River catchment include state forests, agricultural land, national parks, coal mining and urbanised areas. The catchment is typified by narrow valleys to the north that widen to the mid and lower regions. The catchment is valued for its rich biodiversity and diverse landforms (Great Lakes Council 2014). Adjacent to the Karuah River catchment, the Myall Lakes catchment has an area of more than 400 km2. The Myall River is the major tributary of this catchment, with its headwaters extending to Craven Nature Reserve and the Kyle Range. Land use within the Myall Lakes catchment ranges from agricultural, with livestock farming being most popular, to forestry and protected areas on steeper slopes and small urban and peri-urban areas such as the townships of Bulahdelah, as well as the popular tourist destinations of the Tea Gardens-

Hawks Nest area. Much of the native woodland cover within the two catchments remains intact. Cleared areas exist in the valleys to the north and east and towards the coast in the south of the two catchments. Here, native trees remain as scattered or paddock trees standing above managed pastures of native grass (Great Lakes Council 2015). Scattered tree species common within the catchments include eucalypts such as Tallowwood, and several species of Gum and Mahogany trees. Several Angophora spp. and Corymbia spp. are also common, such as the Smooth and Rough-barked Apple, and Bloodwood and Spotted Gum, respectively (MidCoast Council 2018).

3.2.2 Modelling fine-scale connectivity

This study follows a framework described by Lechner et al. (2015a, b) to characterise connectivity based on fine-scale dispersal behaviour (Figure 3.1). The framework has the following workflow:

(1) Identification of key ecological connectivity parameters

(2) Pre-processing spatial data based on these parameters

(3) Inputting spatial data to existing connectivity modelling software

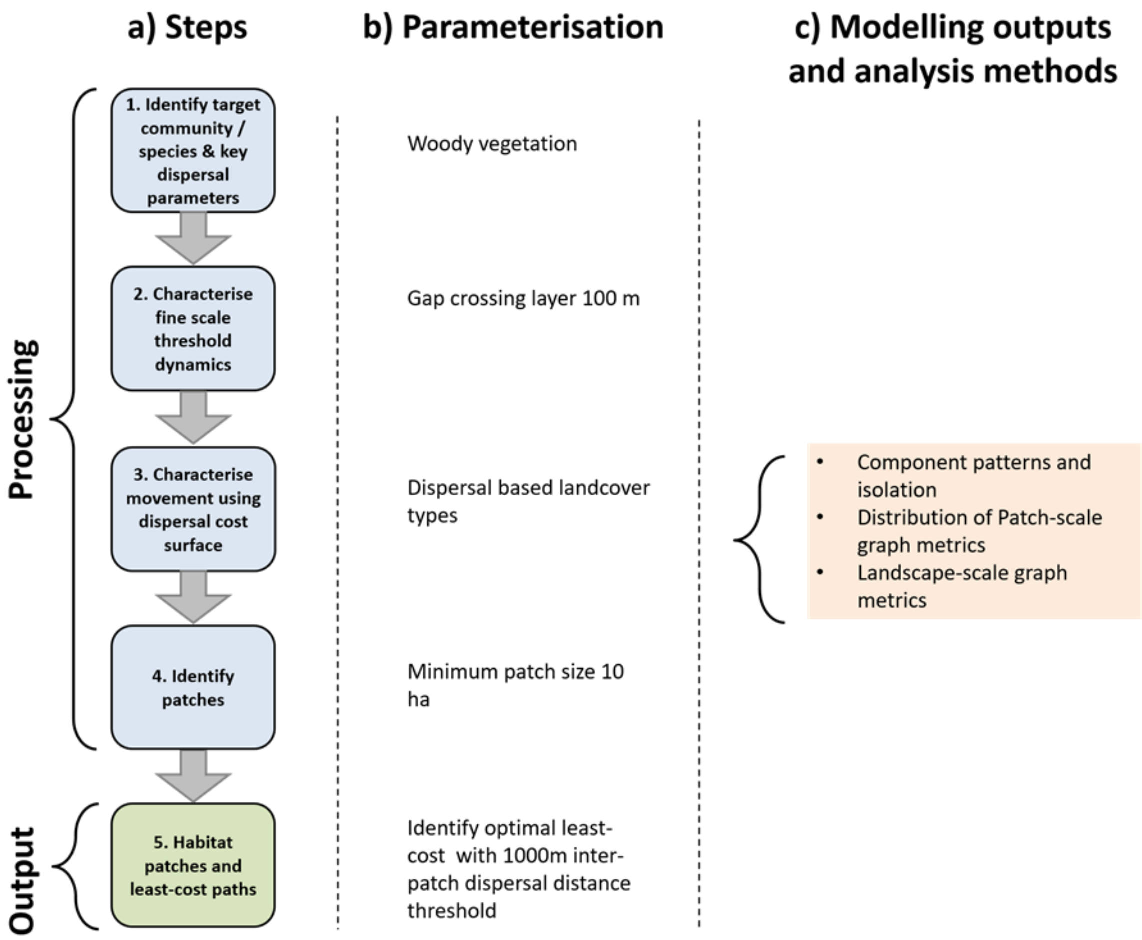

Figure 3.1. Flow diagram describing the workflow of connectivity assessment of the Karuah-Myall catchments.

This study modelled generic connectivity between environmentally similar habitats instead of a particular species, comparable to the land-facet approach that has been applied in Australia (Lechner et al. 2015a, b, c) and internationally (e.g., Brost and Beier 2012). A modelling approach such as this is a balance between the complexity of parameterising and interpreting a multi-species connectivity model, and the simplicity of a structural connectivity model that ignores the inter-species complexity of movement between patches (Lechner et al. 2015b). This approach is based on a "general representative species" which considers native woody vegetation as habitat; it has been applied previously to similar woodland dominated landscapes in Australia (Lechner et al. 2015b, c). Such patches are habitat for the majority of the native fauna in the region affected by fragmentation, as well as floral species that rely on them for dispersal (Lechner et al. 2015b).

3.2.2.1 Identification of key ecological connectivity parameters

The model was parameterised using dispersal values from a systematic review by Doerr et al. (2014). This review assessed how structural connectivity facilitates dispersal and synthesised average values for gap-crossing distance and interpatch dispersal distance thresholds from 80 studies from 98 sources. These values were relevant to the present study as most of the reviewed studies have similar ecosystems and are also impacted by fragmentation from agriculture. The connectivity parameters were:

(1) 1000 m interpatch dispersal distance

(2) 100 m gap-crossing distance

(3) 10 ha minimum habitat patch size

3.2.2.2 Pre-processing spatial data based on these parameters

Along with these ecological parameters, three spatial inputs were used: a habitat patch layer, a dispersal cost surface based on land use mapping and a gap crossing layer based on the gap crossing distance threshold.

3.2.2.2.1 Creation of habitat map and land use resistance surface

Land use and land cover maps were provided by MidCoast Council (2018), New South Wales, Australia. A general land use map and two vegetation land cover maps were used to derive the necessary spatial inputs. We manually edited the spatial data to prepare for the modelling process as the original data were considered inadequate for modelling fine-scale connectivity; this editing is detailed below. The final habitat map is made up of combined native woody vegetation features, whereas the dispersal cost map was a composite of broad land cover classes (Table 3.1) which includes native vegetation and the gap crossing surface. All data processing was performed using ArcGIS software (ESRI 2018).

Table 3.1: Description of original land use zones, conversion to broad resistance classes and resistance values. *Note the original land use layer was updated manually and with Open Street Map data as described in the text.*

Original land use	**General class**	**Resistance class**	**Resistance percentage**	**Resistance pixel value**
Sclerophyll shrubland, sclerophyll forest, coastal dry forest, coastal headland, woodland, dry heathland, wet heathland, mangrove woodland, dry rainforest, riparian forest, tall shrubland.	Habitat			
Sand complex, grassland, sedgeland, rushland, freshwater meadow mix.	Non-habitat	Other	100%	5
Cleared, golf courses, parkland, parkland/grassland, residual pine forest, cleared pasture, managed pine plantation, rock, sand, cleared grassland.	Other			
Urban or residential development, quarry, mining strip, industrial land, landfills, schools, mines-coal, fences.	Infrastructure	Infrastructure	200%	10
Bridleway, construction road, glc road, motorway, motorway link, rail, residential road, rest areas, secondary roads, service roads, tertiary roads, tertiary link, tracks.	Roads	Roads	200%	10
Water, river, stream	Water	Hydro	300%	15

The primary land cover datasets provided by MidCoast Council had three different representations of the distribution of land cover in the study area. These datasets were:

(1) MidCoast Council compiled fine-scale vegetation community map

(2) Mid North Coast Forest ecosystem distribution map

(3) Great Lakes Council native vegetation distribution map

The layers were overlaid and manually corrected while referring to an ArcGIS base map (ESRI 2018). The goal was to produce a harmonised vegetation layer by combining the most accurate components of three primary datasets. This resulted in a composite vegetation layer that was spatially complete and correct for the year 2016.

Roads and waterways were merged with both the habitat and land cover maps to provide a better representation of vegetation and land use patterns which potentially impact fine-scale movement. For example, by adding roads, we were able to identify discontinuities in vegetation patches which were originally mapped as a single patch. We used Open Street Map (OSM) data to identify roads, streams and rivers that were missing from the existing datasets. The following edits were made to the data from OSM:

(1) Filter out small roads and tracks if they did not show up on Google Earth/Google Maps. These were dirt roads or abandoned roads that had little to no traffic and thus do not affect movement.
(2) Buffer the roads and waterways to a total width of 7.5 m. This ensures that the roads were wide enough to show up as continuous strips with no breaks.
(3) Combine roads with habitat map and the subsequent land use map.

To finalise the habitat map, we manually digitised missing vegetation areas large enough to be considered as a patch (≥ 10 ha).

Resistance to dispersal can be described as how movement costs associated with different land cover reduces the maximum distance individuals can travel. For instance, a land cover which doubles dispersal cost would reduce the interpatch dispersal distance threshold of 1 km to 500 m. This study follows the same dispersal costs assigned to each pixel as in Lechner and Lefroy (2014).

To produce a land use resistance map, the classes from the original data sets were categorised into broader classes based on the general ways in which land cover affects movement (Table 3.1). Dispersal costs varied from no cost (habitat and non-habitat) to water which reduced distance by 300%. General classes were further grouped into resistance classes and given a resistance pixel value that represents the cost of movement. Resistance pixel values used are multiples 5, which is the pixel size of all spatial data used in this study. A resistance value of 5 will have no cost to movement and pixels with a resistance value of 10 will have twice the resistance and 15 with three times the resistance.

3.2.2.2.2 Gap-crossing layer

The gap-crossing layer identifies distances between structural connectivity elements and patches beyond the gap-crossing distance threshold. Areas beyond this threshold act as barriers to dispersal and vegetated areas smaller than the minimum patch size are considered as structural connectivity elements. In addition, we manually digitised 14,125 trees (points) and 6703 groups of trees (polygons) which were not included in the original land cover maps. These were trees within cleared or pastural land. A final land cover map consisting of the land cover classes and scattered trees was produced that depicts general land cover classes found within the study area (Table 3.1). The gap crossing layer was created by combining the additional trees with the habitat map and applying a buffer of half the gap-crossing distance threshold (50 m) to all vegetation. Areas outside the buffer distance are considered as barriers to movement. If connectivity elements are present within the gap-crossing distance, the buffers will meet or overlap, allowing for movement. Dispersal will not happen outside the buffered areas.

3.2.2.2.3 Dispersal resistance layer

The dispersal resistance surface describes how land cover between patches restricts movement. It is produced by combining the land use map and the gap crossing layer. The gap-crossing layer takes priority over other land cover classes. The resulting dispersal cost layer is one that acknowledges fine-scale threshold dynamics as it ensures dispersal is impossible where gaps are greater than the gap-crossing distance. The layer also allows for modelling of cumulative costs, where dispersal is possible but may be impeded by land use.

3.2.2.3Connectivity modelling

A graph theoretic approach along with least-cost paths was used to assess connectivity across the two catchments. Using a graph theoretic approach, we were able to characterise the landscape as a network of patches connected by links, described by least-cost paths (Minor and Urban 2008; Dale and Fortin 2010). We modelled connectivity using Graphab 2.2 software (Foltête et al. 2012). The outputs from the connectivity model were interpreted by visualising fragmentation and least-cost paths and quantifying the importance of patches and linkages using graph metrics (Figure 3.2).

In the first stage of analysis, we identified patches or groups of interconnected patches that are isolated from other patches, known as "components". Their boundaries are identified by Graphab, at the midpoint between patches from different components, and are used for visualisation purposes only (see Figure 3.3). Spatial patterns of these components are useful for characterising fragmentation and barriers to connectivity at the regional scale (Lechner et al. 2015c). Large components describe multiple patches that are connected, and help with the characterisation of how regions are connected. Numerous small components represented by a single or several small patches describe regions where dispersal is highly constrained.

At the next stage, graph metrics were used to assess the significance of patches and links within the connectivity network. We calculated two patch scale graph metrics to characterise the importance of patches and linkages for contributing to dispersal. These were the delta integral index of connectivity (dIIC) (Pascual-Hortal and Saura 2006; Saura and Pascual-Hortal 2007) and clustering coefficient (CC) (Ricotta et al. 2000; Minor and Urban 2008). dIIC describes the impact of the loss of habitat availability caused by the removal of the focal patch relative to the connectivity network. The higher the value, the higher the connectivity. CC measures path redundancy between the patch and its neighbouring patches. A higher coefficient means alternative pathways exist for linking neighbouring patches. This is visualised by Graphab as nodes and links across a network.

We assessed the contribution of scattered trees to fine-scale connectivity by modelling connectivity within the study area for two scenarios. The default scenario uses a dispersal cost map that obeys gap crossing distance thresholds through the gap crossing layer, which includes structural connectivity elements such as scattered trees. In the second scenario, we modelled connectivity with a dispersal cost map that is not limited to movement beyond the gap crossing distance threshold. By comparing the two scenarios, we were able to highlight the contribution of small patches and scattered trees to fine-scale connectivity; more specifically, a comparison focused on relative movement patterns of least-cost paths, and the distribution of component boundaries, nodes and links across the landscape can be made.

Finally, we tested the sensitivity of the model to dispersal costs and patch size at the landscape-scale using graph metrics. We repeated the modelling and analyses with another scenario that used only the habitat map without dispersal costs. This allowed for the identification of key dispersal distances for connecting the catchment. It also functioned as a sensitivity analysis, characterising how the interpatch dispersal distance effected the results of the analysis. We also modelled the default scenario with varying minimum patch sizes, ranging from 1 to 30 ha, to determine the influence of patch size on the results. In addition, patch size may decrease or increase the probability of accurately mapping and extracting these patches (Lechner et al. 2009). For each scenario, we calculated the landscape-scale metrics, number of components (NC) and the integral index of connectivity (IIC), to assess overall differences in connectivity patterns. The IIC calculates the probability of two randomly placed dispersers accessing each other (Pascual-Hortal and Saura 2006) and NC is a simple measure of the number of isolated patches in the landscape (Urban and Keitt 2001).

3.2.3 Connectivity network protection status and contribution to Great Eastern Ranges

To assess how important existing national parks, forest reserves and other protected areas are for connecting patches, we overlaid the connectivity modelling outputs with protected area spatial data. A single protected area spatial dataset was produced, consisting of the following classes: National Parks, Forest Reserves, Coastal Wetland, Environmental Conservation, Environmental Management, Flora Reserve, Forestry, Protected Area, State Conservation Area and State Forest. We then identified whether patches and links which had no protection status were important for connectivity within the study area.

We also assessed visually how the Karuah-Myall catchments contribute to connectivity across the Great Eastern Ranges (GER), a national scale regional planning and connectivity initiative centred on the Great Dividing Range and the Great Escarpment. The GER spans the Grampians, Western Victoria and Far North Queensland (https://www.ger.org.au), and crosses the Karuah River catchment to the west.

3.3 Results

3.3.1 Least-cost paths and components

A visual assessment of Figure 3.3a shows that the Karuah-Myall catchments are generally well connected: almost all habitat patches within the landscape are linked to each other, except for four isolated patches in the south-east, as denoted by the component boundaries. These components are visualised in Figure 3.3a as patches surrounded by purple lines. The occurrence of least-cost paths between patches (red lines) indicates that the cumulative cost-weighted distance between patches was less than 1000 m and also that the gap-crossing distance between structural connectivity elements was less than 100 m. Examples of the least-cost paths are shown by the insets. Least-cost paths avoid high resistance land covers such as settlements.

3.3.2 Patch-scale graph metric—delta integral index of connectivity

The dIIC highlights patches based on their potential to facilitate dispersal and their total area, as well as important linkages (Figure 3.4). Figure 3.4a shows a uniformly distributed network of patches with disproportionately higher dIIC values, due to their contribution to connectivity within the landscape. A distinct spine of high dIIC value patches and linkages extends from the north to the south-east and then to the south. This spline branches into two at the central region where one branch continues south-west and the other south-east. These are mostly large sized patches, with the high dIIC linkages between them being critical for connecting the catchments.

3.3.3 Importance of scattered trees and small patches

We modelled connectivity for the same landscape without incorporating gap crossing layer (no scattered trees) to illustrate the impact of scattered trees on connectivity. Figure 3.3b, Figure 3.4b and Figure 3.5b show the movement patterns and distribution of key nodes and linkages in a landscape where movement is allowed beyond the gap crossing threshold. Similar to Figure 3.3a, Figure 3.3b shows a generally well-connected landscape; there is only one isolated patch in the south-east. The cluster of patches that was previously isolated in the southernmost tip in the east is now connected to its surrounding patches. Least-cost paths appear more frequently in this scenario (blue lines), and this is apparent in Figure 3.3d and f. By comparing Figure 3.3c and d, we are able to visualise the effect of a gap crossing distance threshold on movement. Without the threshold, least-cost paths extend beyond 100 m between gaps, ignoring threshold dynamics and cumulative costs to movement. Figure 3.3e shows how movement of the least-cost paths are sensible and utilises scattered trees as stepping stones to a nearby patch; this contrasts with Figure 3.3f in which they are able to cross the open matrix while ignoring scattered trees.

Figure 3.4b shows a similar distribution of dIIC values to the default scenario, where a uniformly distributed network of patches with disproportionately higher dIIC values exists. An almost identical spine of high dIIC value patches and linkages is also present here. The upper limit for dIIC for this scenario is lower than the default scenario.

The CC highlights patches with low redundancy and allows us to address the effect of paddock trees on the role of patches. A high CC value indicates that there are alternative pathways to reach neighbouring patches. The two scenarios are compared in Figure 3.5. In Figure 3.5a, there are many nodes with low CC values and many of these occur along the strips of cleared land in the west, south and east. Patches with low CC values indicate that they are crucial to connectivity as they provide a unique link to other patches. The same trend is seen in the scenario without the gap crossing layer (Figure 3.5b), but with patches having generally higher CC values. Again, insets are included to clarify movement patterns. Figure 3.5d, f shows that more patches are connected when the gap crossing threshold is not considered. The component boundary that exists in Figure 3.5c is now gone in Figure 3.5d. Without a threshold where movement can only occur if two scattered trees are close enough, least-cost paths appeared across a wide river with no stepping stone structures in between, to connect neighbouring patches. Figure 3.5c, e shows that, when structural connectivity elements are considered, patches generally have a lower redundancy value compared to a scenario where they are ignored. Again, if scattered trees are ignored patches are connected to neighbours by multiple routes, increasing their redundancy (Figure 3.5d and f).

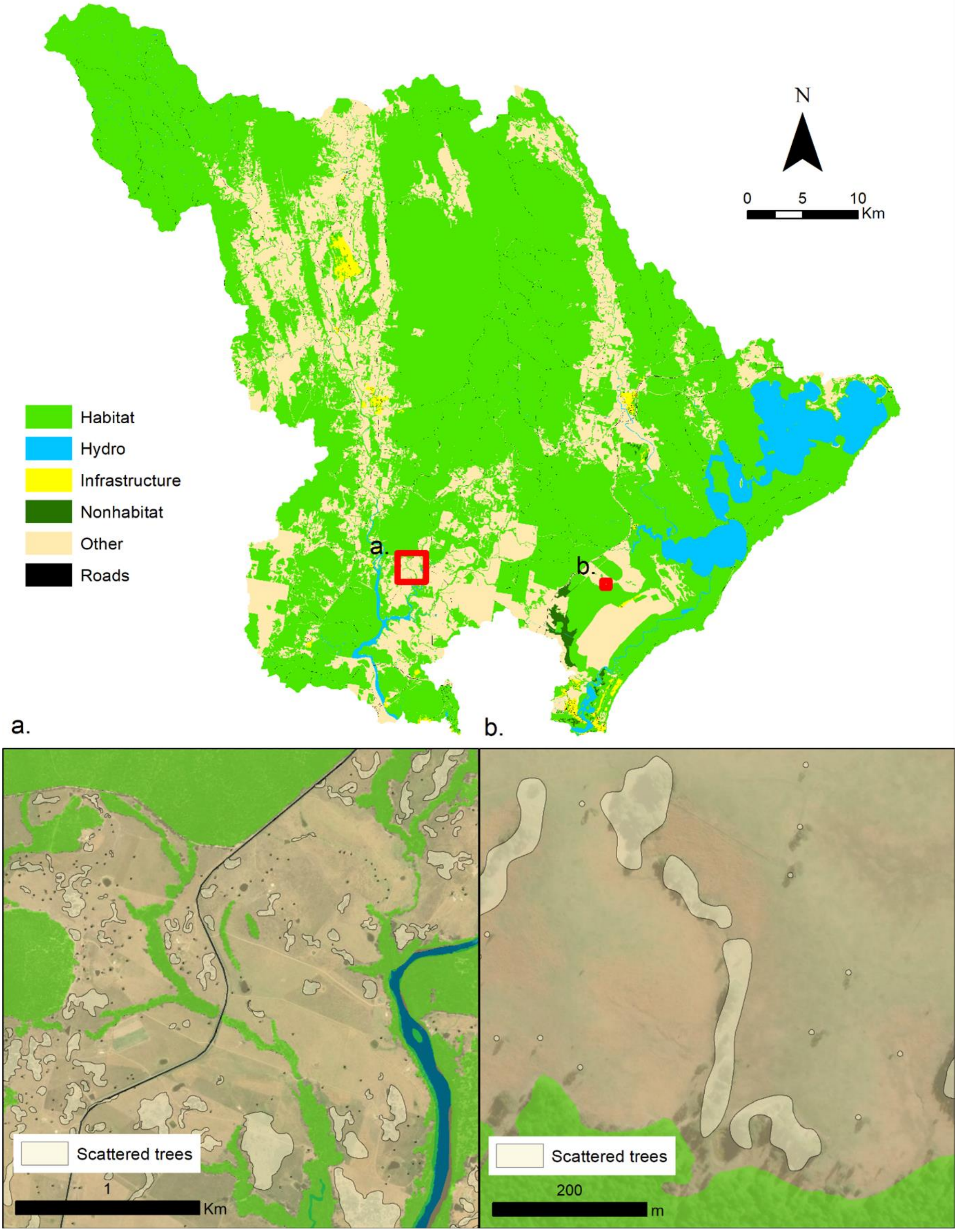

Figure 3.2: Land cover map depicting general land cover classes (Table 3.1) and scattered trees. *Individual scattered trees are shown as points and grouped scattered trees as polygons in insets a, b.*

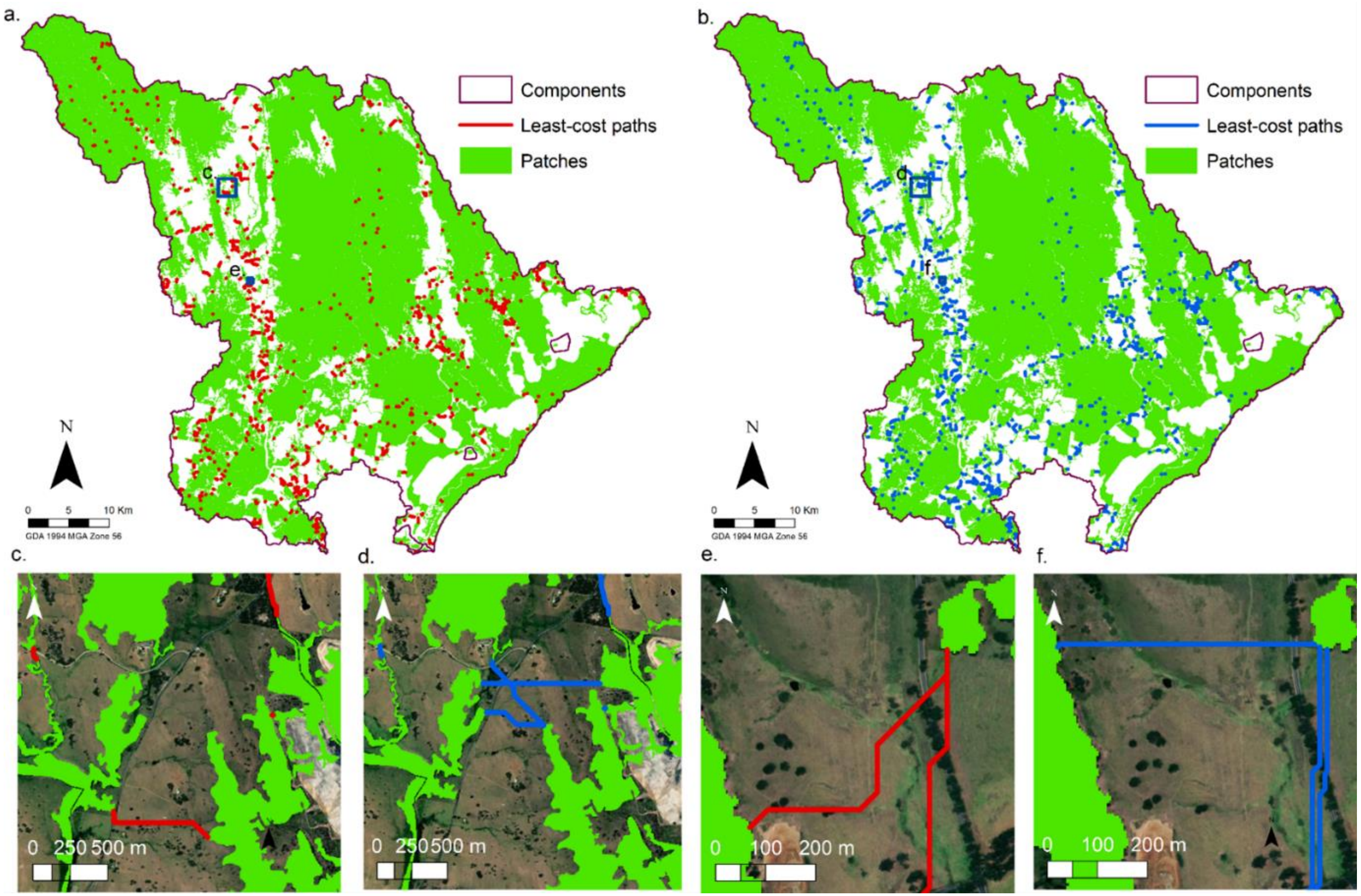

Figure 3.3: Habitat patches, least-cost paths, and component boundaries of the Karuah-Myall catchments. *A, c, e has the gap crossing layer characterising movement between scattered trees and b, d, f shows movement patterns without the gap crossing layer. Insets c-f shows the characteristics of the least-cost paths.*

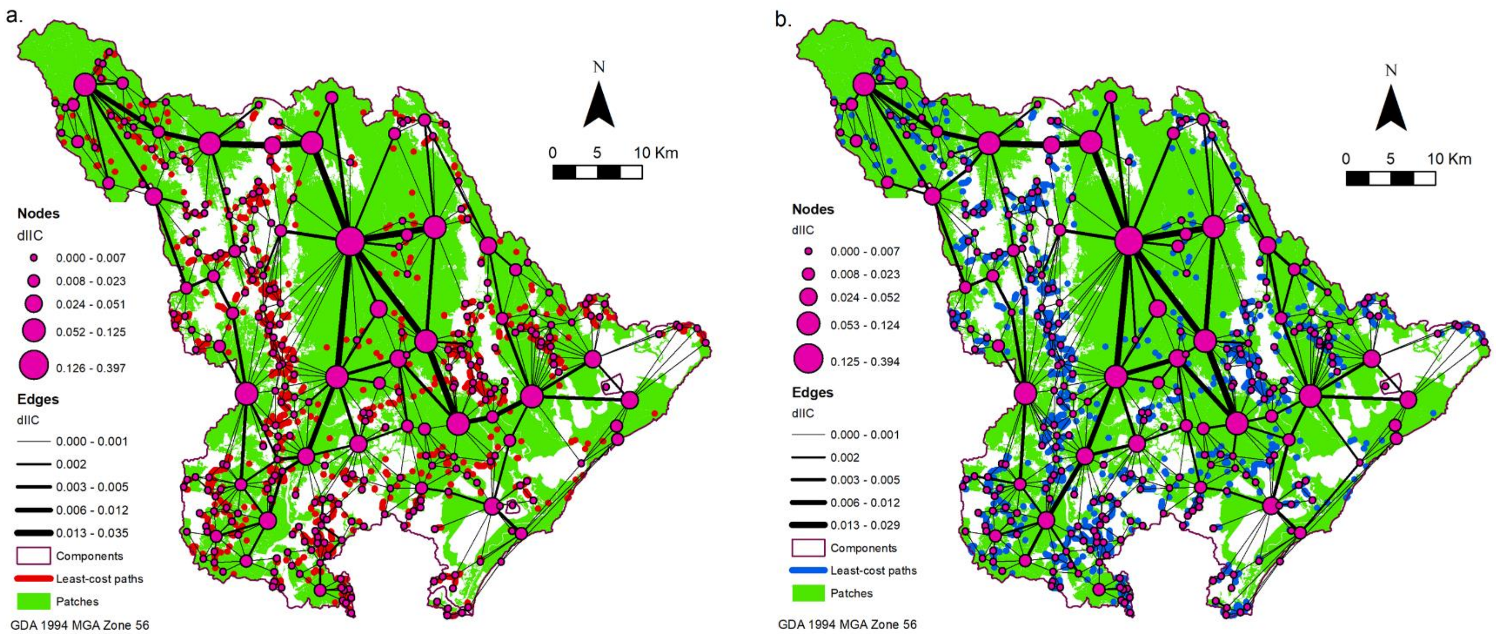

Figure 3.4: Patch-scale delta integral index of connectivity (IIC) metric modelled. *a contains the gap crossing layer characterising movement between scattered trees and b without. The importance of linkages and patches is denoted by the thickness of the lines and size of the circles respectively. The circles are located at the centroid of each patch.*

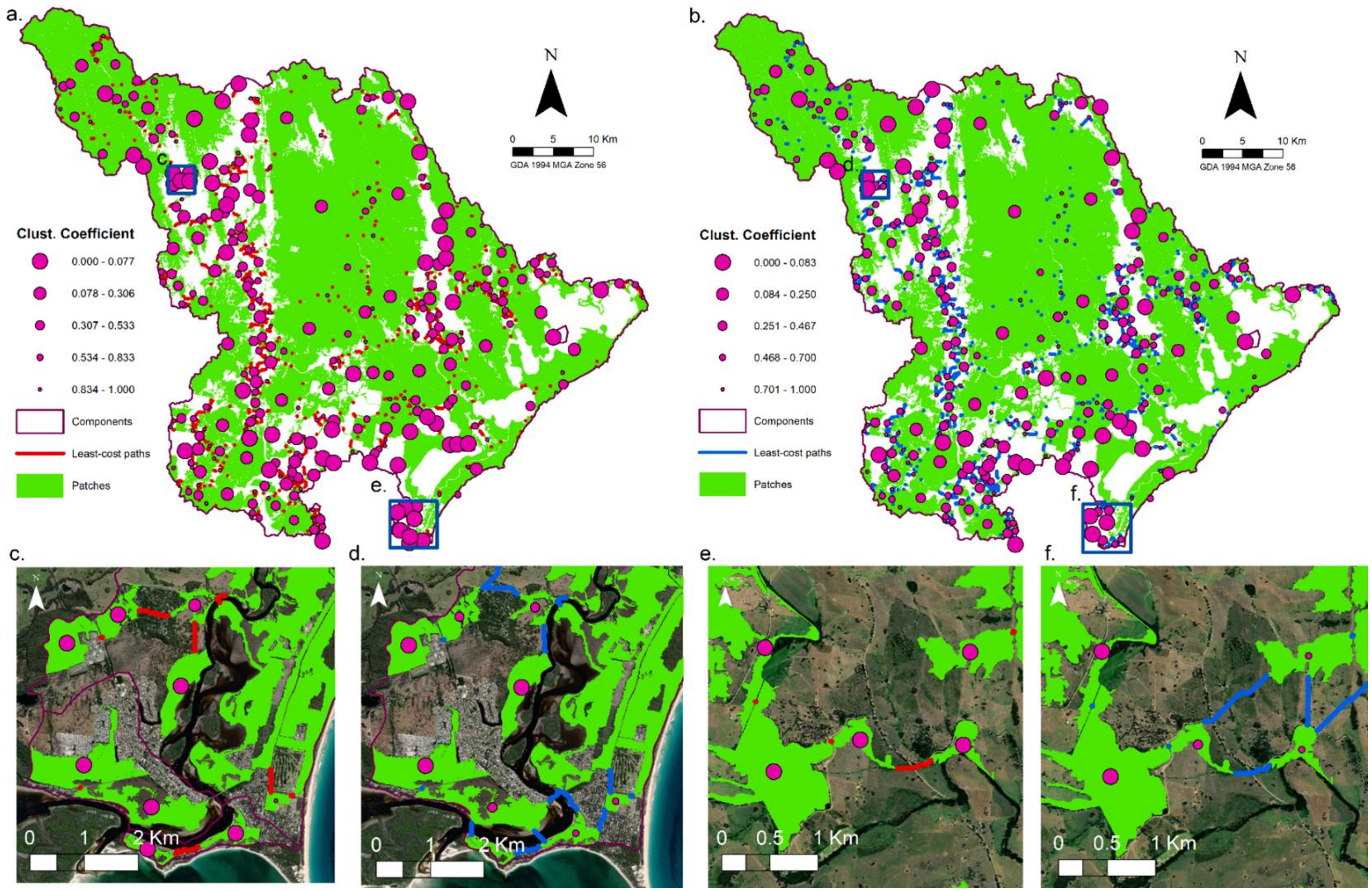

Figure 3.5: Patch-scale metric, clustering coefficient (CC), characterises the level of redundancy (i.e., alternative routes) between neighbouring patches. *a, c, e has the gap crossing layer characterising movement between scattered trees and b, d, f does not. Large circles represent crucial patches for connectivity. Insets c-f shows the characteristics of the least-cost paths.*

3.3.4 Sensitivity analysis

Figure 3.6 shows that there was very little difference in global connectivity (i.e., at the regional scale) between the different choices in model parameterisation. This indicates that the greatest driver of connectivity within the Karuah-Myall catchments is interpatch dispersal distance, not resistance due to land cover. The sensitivity analysis also provides a coarse-scale assessment of connectivity for species with shorter and longer dispersal distances than the general representative species which was the focus of our study. There was a large decrease in NC and increase in IIC at the 50 to100 m distance threshold for all parameterizations, suggesting that species with these movement distances or less are most likely to be affected by fragmentation in the catchments. However, these species, which tend to be small sized, will have lower requirements for total patch area so are less likely to be impacted by fragmentation. Figure 3.6 also shows that scenarios with and without gap crossing are very similar. In Figure 3.6a, there are only two components at the 1000 m threshold mark, whereas if paddock trees are considered, the number of components increases to five. Nevertheless, we did not expect a major difference between the default scenario and one without gap crossing. On a fine scale, movement patterns and patch redundancy can still be misrepresented if scattered trees are ignored, even when the two scenarios share similar qualities in the distribution of landscape scale metrics. Additionally, Figure 6 shows that NC and IIC decrease and increase with minimum patch size respectively. Analyses with a minimum patch size of 1 ha would require datasets with a spatial resolution of 100 × 100 m or finer which are provided by most conventional satellite used in land cover mapping, such as Landsat (30 m) and Sentinel 2 (10 m). This means that satellite spatial resolution is unlikely to impact the delineation of habitat patches (Lechner et al. 2009).

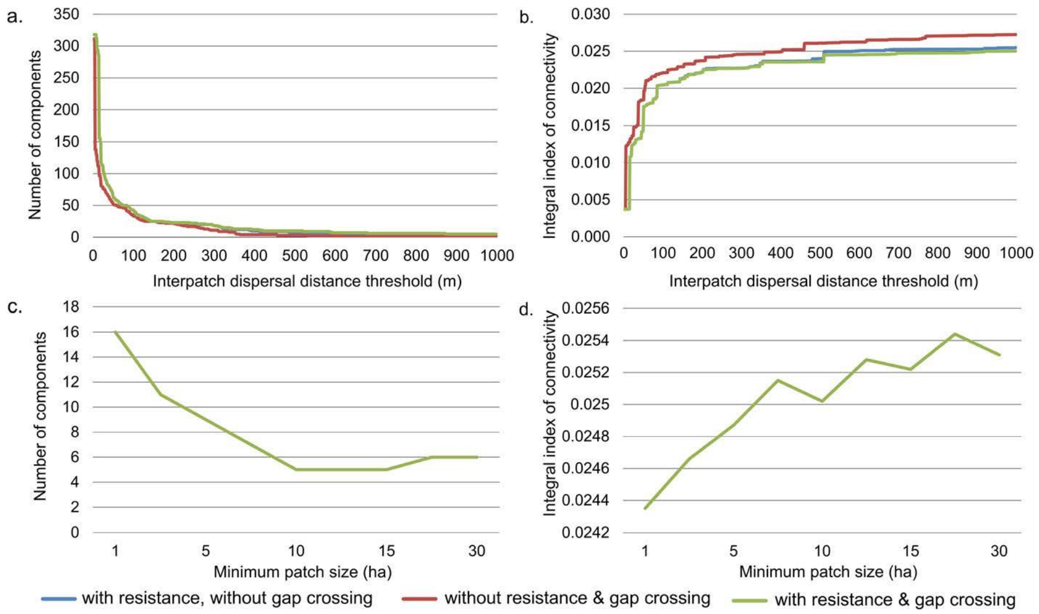

Figure 3.6: Number of components and integral index of connectivity versus interpatch dispersal distance threshold. *a, b and minimum patch size c, d. a, b compare model scenarios with resistance, without resistance and without the gap crossing layer, while c, d connectivity shows that the metrics are negatively and positively correlated to minimum patch size respectively.*

3.3.5 Connectivity in protected areas and contribution to Great Eastern Ranges

Figure 3.7 shows dIIC values and indicates that the majority of high dIIC nodes and linkages are within protected areas. The exception is a region to the north-west which has no protection status (Figure 3.7a). This area includes high value patches and linkages in the north which connects west and east of the study area. Another key region without protection is in the south-west (Figure 3.7b).

Figure 3.8 shows dIIC values and the overlap with the Great Eastern Ranges. There are few nodes and linkages with high dIIC values within the GER. There is one significant node to the north and one more just below the middle, in the west. These areas are important for connectivity.

Figure 3.9 shows the Karuah-Myall catchments in the context of the GER. Figure 3.9a shows the location of Karuah-Myall catchments in relation to the GER at the national scale. Figure 3.9b shows visually that there is a cleared region between north and south (red arrow) forested areas in the GER to the west of the Karuah-Myall catchments. The yellow arrow in Figure 3.9b represents a hypothetical potential linkage enabling movement from patches in the north to patches in the south of GER. The Karuah-Myall catchments is part of a region close to the east coast which potentially also provides another north to south linkage. In Figure 3.9c, the arrows are used to visualise hypothetically how the Karuah-Myall catchments connect to the GER in the north-west, and the south-west, shown by the light blue and dark blue arrows respectively.

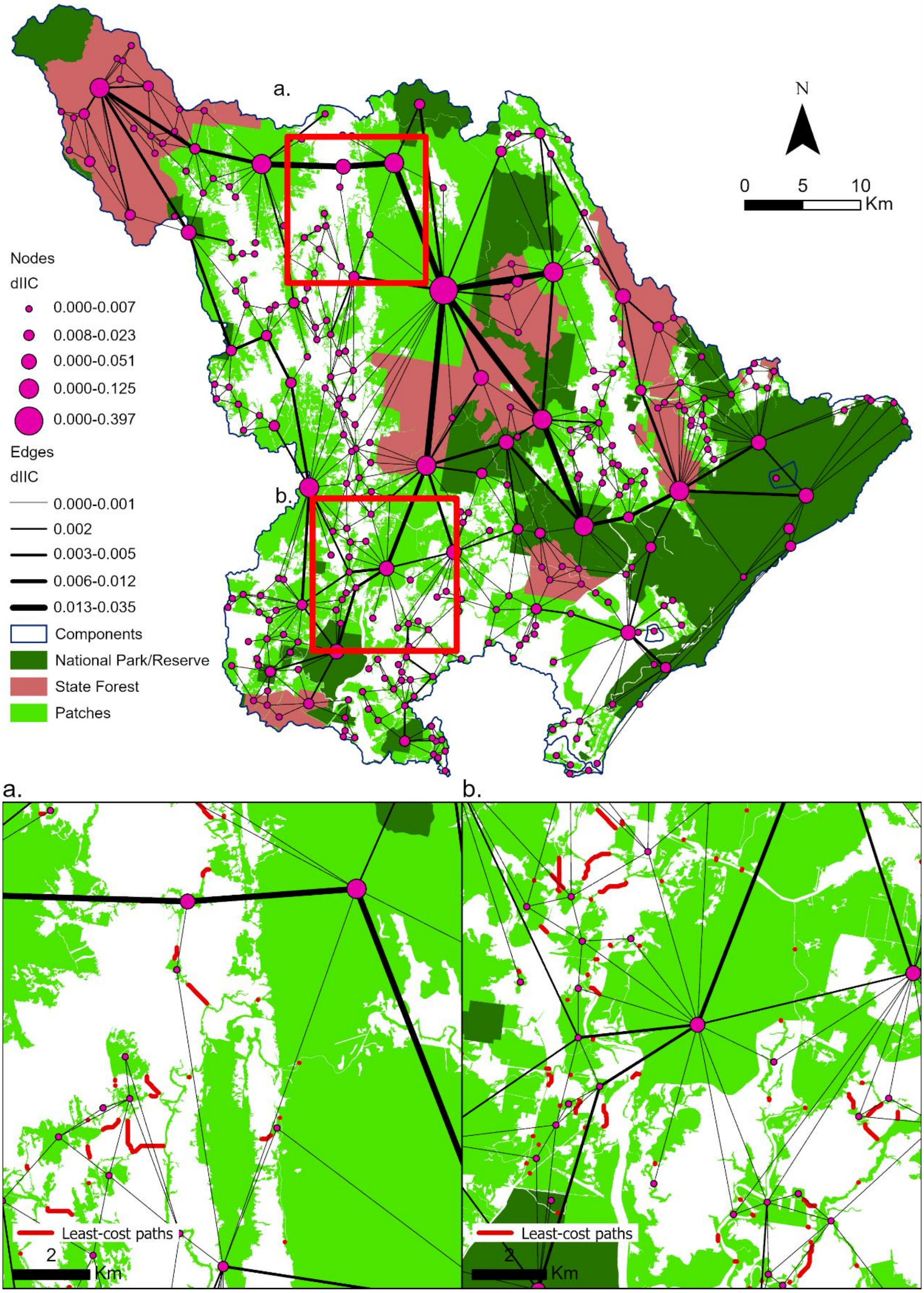

Figure 3.7: Habitat patches and protected areas with component boundaries and delta integral index of connectivity (dIIC) for patches and linkages. *The importance of linkages and patches is denoted by the thickness of the lines and size of the circles respectively. The circles are located at the centroid of each patch.*

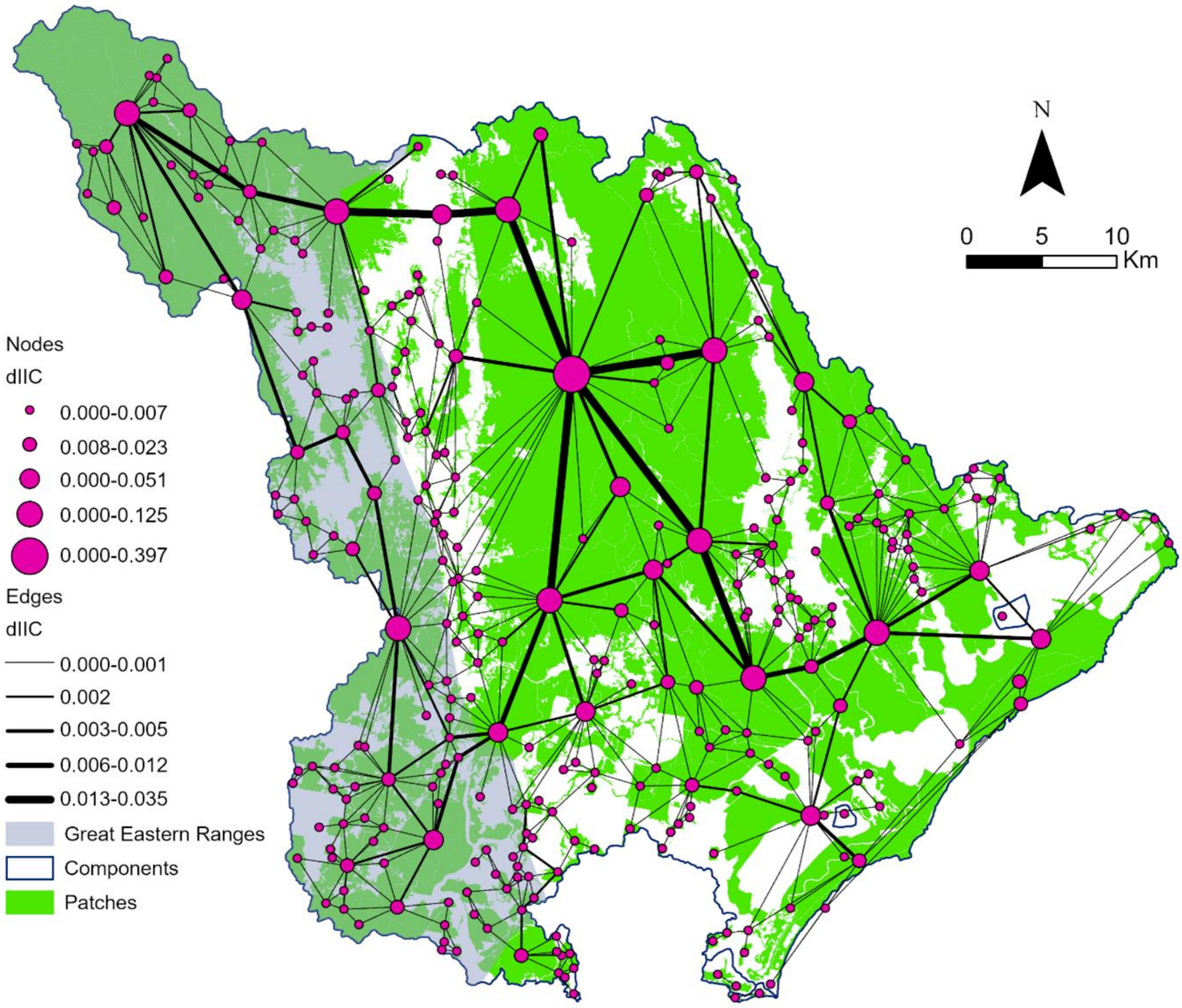

Figure 3.8: Habitat patches and the Great Eastern Ranges (GER) with component boundaries and delta integral index of connectivity for patches and linkages. *Important linkages and patches are denoted by thick lines and circles respectively. The circles located at the centroid of each patch describe patch-scale graph metric values.*

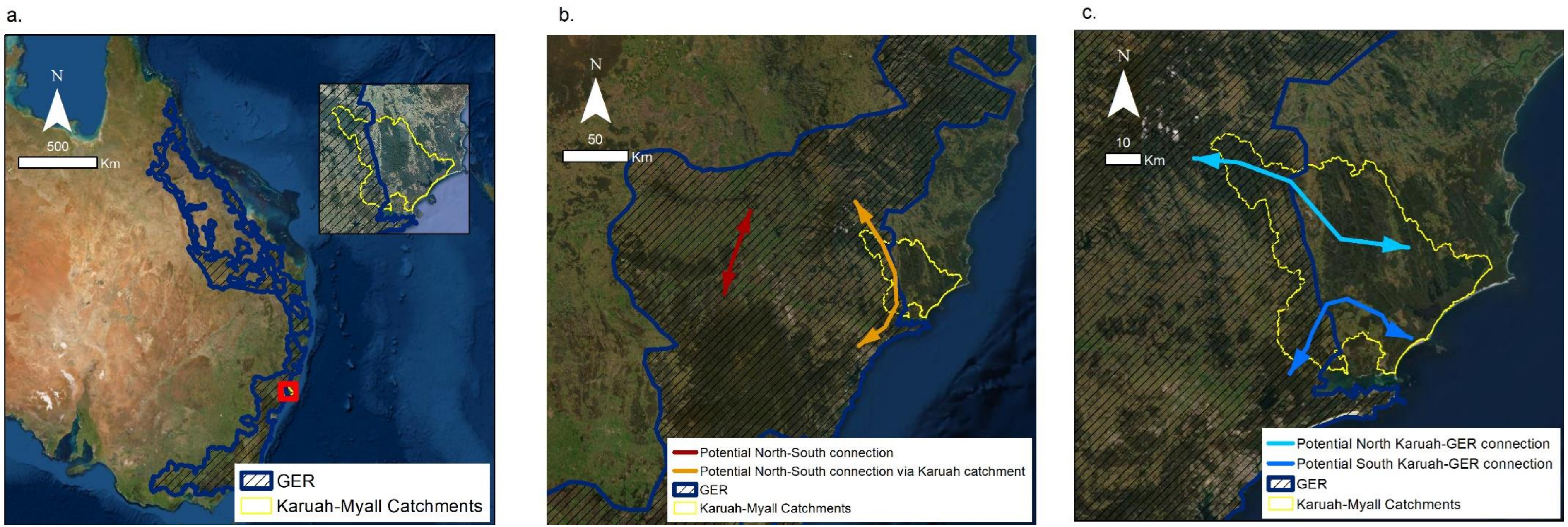

Figure 3.9: a. Extent of the Great Eastern Ranges and the location of the Karuah-Myall catchments. *b. Hypothetical North-South connection between the Karuah-Myall catchments and the GER. c. Hypothetical pathways denoted by arrows showing how Karuah-Myall catchments supports movement between coastal patches within the study area and the GER.*

3.4 Discussion

3.4.1 Overall connectivity

Our study showed that the Karuah-Myall catchments are well connected, with only four isolated patches evident from the results of the analyses. The study area is fragmented by two agricultural regions along the valley floors. Although these areas have been cleared, they still support many scattered trees spaced below the gap-crossing threshold distance. The sensitivity analysis shows that species with an interpatch dispersal distance threshold of 50 to 100 m or less are likely to be mostly affected by fragmentation in the study area (Figure 3.6). This study provided a coarse level general assessment of connectivity for the Karuah-Myall catchments. While we only used a "general representative species" for the parameterisation of the model, the sensitivity analysis suggests that it is likely that the catchments are well connected for most species that depend on woody vegetation. Further assessments for species of conservation concern which have more specific habitat requirements (e.g., utilise a subset of woody vegetation, or perhaps grasslands) and or have specific movement requirements not captured by our resistance model (i.e. roads are barriers to movement) are potentially required.

3.4.2 Importance of scattered trees and small patches for representing fine scale dispersal patterns

Many field-based studies have shown that gaps in discontinuous habitats limit movement across fragmented landscapes (Desrochers and Hannon 1997; Rail et al. 1997; Grubb and Doherty 1999; Bélisle and Desrochers 2002; Creegan and Osborne 2005). The presence of scattered trees and the distance between them can impact overall connectivity within fragmented landscapes, as many species avoid being exposed in the open matrix. For example, Squirrel Gliders (*Petaurus norfolcensis*) have been observed to glide between individual trees not more than 75 m apart (van der Ree et al. 2004) and forest birds such as the Grey-shrike Thrush (*Colluricincla harmonica*) and White-throated Treecreeper (*Cormobates leucophaea*) perceive cleared land as barriers to movement and gaps more than 100 m significantly reduce their functional connectivity (Robertson and Radford 2009).

Our study explicitly incorporates the ecology of fine-scale connectivity through mapping vegetation at a high spatial resolution by modelling a gap-crossing distance, building on previous work (Lechner and Lefroy 2014; Lechner et al. 2015b, c) specifically to address the role of scattered trees. This approach is especially important in pasture dominated landscapes which are generally very open, apart from scattered trees. Our results showed that without modelling movement between scattered trees (Figure 3.3b), the least-cost paths were linear (and unnatural), with least-cost paths crossing vast cleared areas. The modelling also portrayed a more connected landscape, which overestimates overall connectivity as shown by the reduced number of components (Figure 3.3b) and more linkages with high dIIC values (Figure 3.4b). Models that fail to include scattered trees risk misrepresenting patterns of connectivity in agricultural landscapes such as the Karuah-Myall catchments.

The least-cost paths in Figure 3.3b illustrated how dispersal patterns modelled without fine-scale connectivity bear little resemblance to what we would expect from the foray search strategy, commonly used by small- and medium-sized mammals and birds (Sun 1997; Koenig et al. 2000; Wiggett and Boag 2011) and regarded as being the preferred strategy for dispersal in fragmented landscapes (Conradt et al. 2003). Foray searchers would regularly return to their starting habitat to reorient and replenish themselves before gradually travelling further distances to reach new habitat (Conradt et al. 2001). Scattered trees and small patches can provide momentary respite in their search for new habitat. In a study on dispersal behaviour of woodland birds, both sedentary and nomadic bird species such as the Eastern Yellow Robin (*Eopsaltria australis*) and the White-plumed Honeyeaters (*Lichenostomus penicillatus*) were observed to use a foray search strategy for dispersal and moved between scattered trees not more than 80 m apart during their exploratory journeys (Doerr et al. 2011).

Aside from scattered trees, small and linear patches also contribute to connectivity within the study area, demonstrated by the high number of small patches with low CC value patches across the landscape in the default scenario (Figure 3.5a). A low CC value indicates that there are no alternative routes to these patches. In some cases, smaller patches can be the only pathway to an otherwise unreachable larger patch. It has been observed more broadly that conservation value decreases as patch size increases and the intactness of the surrounding landscape increases (Wintle et al. 2019).

3.4.3 Informing tree management policies in agricultural landscapes to support conservation

The outcomes of our study are useful for informing scattered tree management by pinpointing key areas to focus on for tree conservation and regeneration efforts. Modelling with a gap crossing threshold effectively filters for trees that contribute to overall connectivity, and those that do not. Our modelling approach can demonstrate that scattered trees have significant value, which is not the case if conservation is purely focused on habitat area or threatened species. Tree recruitment practices can then be carried out in areas between isolated trees to close the gap and also to ensure that ageing trees are being replaced. Different grazing regimes and degrees of land use intensification can also influence the rate of tree regeneration within a pasture. Overgrazing reduces shade and shelter for seedlings (Bergmeier and Roellig 2014), and increased fertilization from land use intensification disrupts the soil nutrient balance that keeps mature trees healthy (Davidson et al. 2007). At the same time, grazing regimes such as a fast-rotational grazing (Longland 2013) can be proposed to enable farmers to remain economically productive and retain tree health and safeguard tree regeneration.

Of great concern for these degraded agricultural landscapes is that seed dispersal is highly reduced due to increasing fragmentation. Seed dispersal is a key ecological process that controls plant population and community persistence (Higgins et al. 2003; Pearson and Dawson 2005). Scattered trees contribute to the regeneration of these woody communities in degraded lands by acting as seedling nucleation foci (Slocum and Horvitz 2000; Zahawi and Augspurger 2006; Kelm et al. 2008), but their numbers are still in decline due to clearing, natural death and lack of regeneration (Gibbons et al. 2008). Fischer et al. (2009) reported that in their 800,000-ha study area of roughly 3 million scattered trees, the chance of regeneration was extremely low, due to conventional livestock grazing and fertilizer use. By extrapolation, they estimated that millions of hectares of south-eastern Australia’s grazing regions will become treeless if conventional management persists. On the other hand, tree recruitment practices often incur large costs as farmers would have to halt grazing temporarily (Kikoti et al. 2015) and set up tree guards (Baumber et al. 2017) until seedlings can withstand grazing.

3.4.4 Priorities for catchment scale connectivity

Our analyses indicate that scattered trees and small patches make important contributions to overall connectivity across and outside the study area. Many parts of the catchments which make important contributions to connectivity have no formal protection status. While the Karuah-Myall catchments appear to be well connected for a cleared pasture dominated agricultural landscape, east-west linkages across the cleared valley floors should be prioritised to preserve connectivity to ensure future connectivity. The analyses suggest that immediate priority focus areas for enhanced connectivity status or function exist at several areas, notably The Glen Nature Reserve west to Avon River State Forest (no protected status), the Karuah National Park north-east to Myall River State Forest (no protected status) and Karuah National Park to Monkerai Nature Reserve (contribution to Great Eastern Ranges Initiative).

The value of the study area for connectivity is not only for biodiversity within the three catchments but beyond the catchments as part of the GER. The Karuah catchment appears to also connect the coastal forested areas within the Myall Lakes catchment to patches in the GER. Critically, this connection is dependent on number of key patches and linkages in the north (Figure 3.7a).

3.4.5 Future research and limitations

There are several areas which future research should target to build on the findings presented here. Firstly, we used a landscape feature approach to model for a “general representative species” to characterise general connectivity. This differs from a multi-species approach (Lechner et al. 2017). Additionally, scattered tree characteristics such as their height, age and canopy size were not considered in our analyses, which can be critical factors for tree use for many species (Dean et al. 1999; Gibbons and Lindenmayer 2002; Leonard Jr. and DeLotelle 2003). Future work could include a dispersal guild approach as an intermediate between single species models and land-facets approach, or a multi-scenario approach to model connectivity for different landscape conditions or species parameters. These options provide generalisability, while also targeting a specific group of species that have overlapping habitats or exploit the same resources (Lechner et al. 2017). Field data describing the composition of scattered tree species and floristic diversity would also complement the modelling presented here.

3.5 Conclusion

This study modelled connectivity for the Karuah-Myall catchments, a forest landscape fragmented by a matrix dominated by pasture agriculture. Our approach allowed for the importance of fine-scale features such as scattered trees to be quantified from the perspective of connectivity. For realistic fine-scale movement patterns to be characterised from an ecological perspective, scattered trees should be incorporated into spatial data and connectivity modelling. This will allow land managers to identify the important conservation values of these features which are often ignored. More specifically for our study area, the modelling showed that the Karuah-Myall catchments are well connected for a "generalised representative species", and that patches within the catchment may make an important contribution to connecting biodiversity beyond the geographical boundaries of the study area due to its location within the Great Eastern Range. However, connectivity even at large scales can potentially be influenced by the presence or absence of even the smallest features such as scattered trees.

Chapter 4 Ecological connectivity in Environmental Impact Assessments: Modelling alternative highway bypass scenarios

Submitted as: Tiang DCF., van der Ree R, McCaffrey, N, Gibbins C, Lechner AM (in review) Ecological connectivity in Environmental Impact Assessments: Modelling alternative highway bypass scenarios, *Impact Assessment and Project Appraisal.*

4.1 Introduction

Linear infrastructure development continues to expand to support increasing human populations and underpin economic growth (Henderson 2003; Perz et al. 2007). Between 2000 and 2013, the length of legally sanctioned roads grew by 12 million km worldwide, and an additional 25 million lane-km of new roads are expected by 2050 (Dulac 2013). This growth of the infrastructure network has major implications for biodiversity as road building is one of the greatest drivers of habitat fragmentation and biodiversity loss across the globe (Laurance et al. 2015; van der Ree et al. 2015; Ng et al. 2020). Roads can fragment habitat and obstruct the movement of individuals and genes within and between populations, and reduce access to resources and mates (Ceia-Hasse et al. 2018). Fragmentation can impact population abundance and persistence, which can cascade through ecological communities, altering patterns of species composition and richness, and ultimately affecting ecosystem function (Fahrig and Rytwinski 2009; Barrientos et al. 2021).

The first step in addressing the potential ecological impacts of new roads is to formally investigate and quantify them during the planning stage (Mörtberg et al. 2007), often in formal environmental impact assessments (EIAs). Despite this requirement, assessments are carried out with varying degrees of success (Wathern 2013) and the lack of quantitative predictions concerning landscape-scale ecological connectivity still persists within EIAs (Deslauriers et al. 2018; Spanowicz and Jaeger 2019). EIAs often only focus on local or site-level impacts within the construction footprint, partial road sections or narrow investigation footprints (Halpern et al. 2008; Halpern and Fujita 2013; Andersen et al. 2015). Impacts on biodiversity can, however, occur across large areas across the landscape (Hawke 2009). This larger spatial scale is not adequately addressed in EIAs, with existing literature often failing to guide or properly delimit the scope of study areas for EIAs so that spill-over effects are ignored (Fischer 2006; Albert et al. 2017). Furthermore, the need for impact mitigation measures such as wildlife crossing structures are often not considered beyond the local site scale.

The need to consider biodiversity within the EIA planning and decision-making process has been emphasised in legal frameworks such as the EU Directives 85/337/EEC and 2001/42/EC, as well as policy papers such as the Convention on Biological Diversity (Karlson et al. 2014). These address the direct, indirect and cumulative effects of development. In recent years, spatial connectivity has emerged as an area concern for understanding transport infrastructure impacts and for undertaking EIAs (Jepson 2016). Transport planners are aided by the new technologies and approaches to identifying potential environmental impacts (González Del Campo and Gazzola 2020), but even so, there is little guidance for conducting analyses on the effects of infrastructure on ecological connectivity and ways to mitigate impacts (Folkeson et al. 2013; Mimet et al. 2016). Wildlife crossing structures are usually built over linear infrastructure such as highways (Mata et al. 2008; Corlatti et al. 2009; Kusak et al. 2009) to connect habitat, and minimise the barrier effect of transport infrastructure for all terrestrial groups of animals (Iuell 2003; Olsson et al. 2008; Glista et al. 2009).

With advancements in ecological modelling and Geographic Information Systems (GIS) there is great potential for incorporating robust quantitative and spatially explicit ecological assessments into EIAs, for (i) defining the scales of environmental impacts beyond the construction footprint, also known as the road effect zone, (ii) quantifying and visualising impact scenarios, (iii) evaluating alternative alignments and mitigation options, and (iv) raising awareness of cross-scale interactions as well as measuring indirect and cumulative consequences (Plante et al. 2018; Quaglietta et al. 2019; González Del Campo and Gazzola 2020). Existing connectivity modelling methods such as least-cost path analysis, circuit theory and graph theory, provide a modern, diverse toolbox to study and quantify different aspects of landscape connectivity (Urban and Keitt 2001; Adriaensen et al. 2003; McRae et al. 2008; Foltête et al. 2012). Scenario-based methods are especially relevant to the EIA process as they can improve information and advice on how alternative infrastructure development effects ecological connectivity, but these are rarely applied (but see Tannier et al. 2016; Huang et al. 2018; Tarabon et al. 2020; Fullman et al. 2021; Sahraoui et al. 2021). In addition, scenario-based analysis of wildlife mitigation measures is critical for assessing the merits of different mitigation options (Gurrutxaga and Saura 2014; Clauzel 2017). Modelling and GIS approaches for assessment of the consequences of alternative road alignment designs and mitigation measures has the potential to provide improved insights into the long-term viability of proposed developments and to support a more robust EIA, especially where road alignments are chosen to reduce their impact on ecological connectivity.

In this study, we applied ecological connectivity modelling to support EIAs by conducting a spatially explicit scenario analysis of alternative road alignments and mitigation options. The study addressed common criticisms of spatially explicit ecological assessments in EIAs (Patterson et al. 2022) such as the use of single species surrogates, their ad-hoc and local nature and the lack of attention to the relationship between landscape-scale functional connectivity and population viability. First we undertake a short review of the literature on existing ecological connectivity approaches used in linear infrastructure impact assessments to provide the context for this paper in this new and emerging area. We then demonstrate the application of a scenario analysis approach using a case study of a town bypass proposal in which we compare impacts of four different bypass alignment options on ecological connectivity to determine the 'least-negative' option. We then quantify the effectiveness of proposed wildlife crossing structures as mitigation measures for the preferred bypass route alignment. We conclude by reflecting on the challenges and considerations for applying such approaches to EIAs.

4.2 Methods

4.2.1 Study area

The town of Beaufort sits within the Central Victorian Uplands (CVU) and the Victorian Volcanic Plains (VVP) bioregions in western Victoria, south-eastern Australia and is characterised by a patchwork of urban, agricultural, native woodland forest, wetlands and areas of linear vegetation along road and rail corridors (WSP 2021). The Western Highway that passes through Beaufort is the primary highway linking Melbourne and Adelaide (Brown 2013). It is one of Victoria's busiest rural highways, with an average of 11,063 vehicles per weekday travelling the road west of the nearby town of Ballarat (Regional Roads Victoria 2022). A bypass around Beaufort has been proposed to improve road safety and remove heavy goods vehicles from the centre of the town (Brown 2013; Regional Roads Victoria 2022). There are four route options being considered (Figure 4.1), each one skirting north of the town and passing through large, forested areas. The wider study area (Figure 4.1) includes the Camp Hill State Forest to the north and Trawalla State Forest to the south.

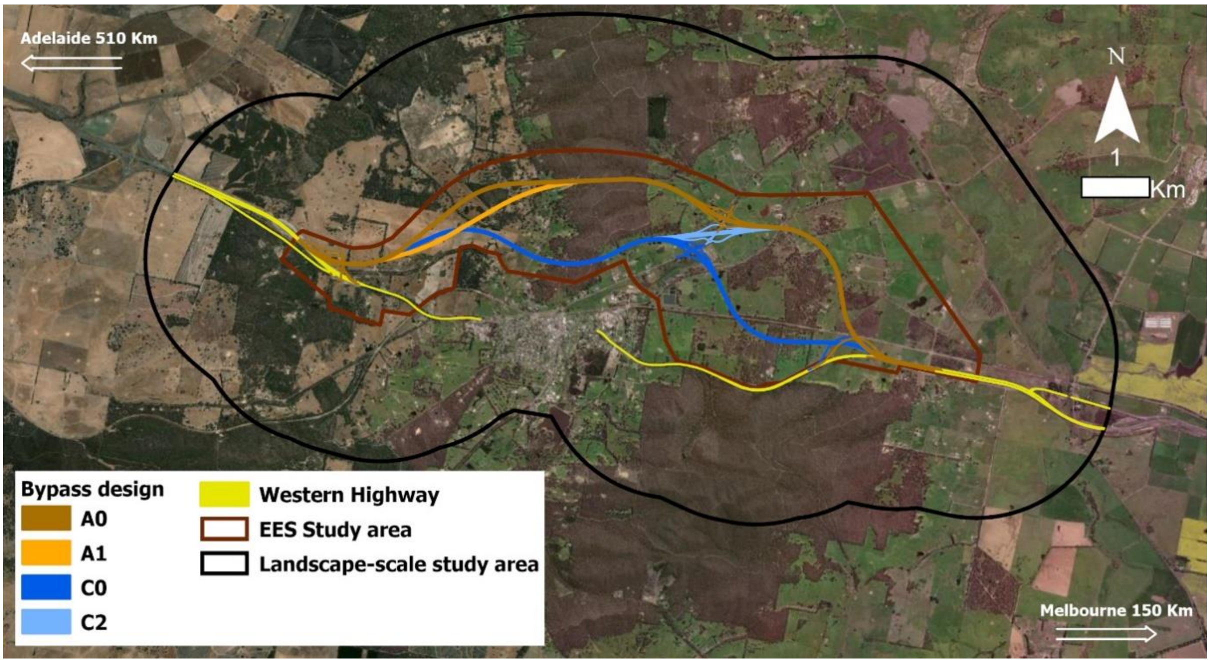

Figure 4.1: Study area for Beaufort Bypass connectivity modelling. *The bypass includes 4 different alignment options (A0, A1, C0 and C2). The EES study area (shown by dark brown line) was extended by 2 km to derive the landscape-scale area used in this study.*

4.2.2 Connectivity assessment overview

In order to assess the landscape-scale impacts of the Beaufort Bypass on ecological connectivity, we extended the area considered in the Beaufort Bypass Environment Effects Statement (EES) (Regional Roads Victoria 2022) to also include the portions of the woodland in the north, southeast and southwest of the town. This was achieved by buffering the EES study area by 2 km, increasing the total area to approximately 80 km^2. A key focus for the maintenance of connectivity in our study area was maintaining north-south links between the native woody vegetation to the north and south of Beaufort.

We applied an expert-based parameterisation of a connectivity model using a combination of least-cost path, circuit theory and graph theory methods. This is adapted from the General Approach to Planning Connectivity from Local Scales to Regional (GAP CLoSR) connectivity modelling framework (see Lechner and Lefroy 2014; Lechner, Doerr, et al. 2015; Lechner et al. 2017 for further details) which describes fine-scale dispersal patterns across large spatial extents. The GAP CLoSR method is based on three key parameters:

(1) The minimum habitat patch size required to support viable populations.
(2) The gap-crossing distance between connectivity elements such as scattered trees, which limit the distances of open ground (gaps) which individuals will move across.
(3) The interpatch-crossing distance threshold, which is the maximum distance an animal is capable of moving between patches of habitat.

We modelled connectivity for four target species and one target species group, each of these representing different species 'guilds' (see Table 4.1). A dispersal guild is defined as a specific group of species that have overlapping habitats or exploit the same resources (see Lechner et al. 2017). A dispersal guild approach allows us to examine the impact of each bypass alignment and the effectiveness of each WCS towards a wide range of species, providing greater generalisability over a single species approach. A multi-scenario approach also allows us to quantitatively identify the best bypass design.

Out study involved a number of processing steps which included the use of remote sensing for land cover mapping and the selection and parameterisation of conservation targets for assessing impacts and mitigation (Figure 4.2); the steps are described in the following section.

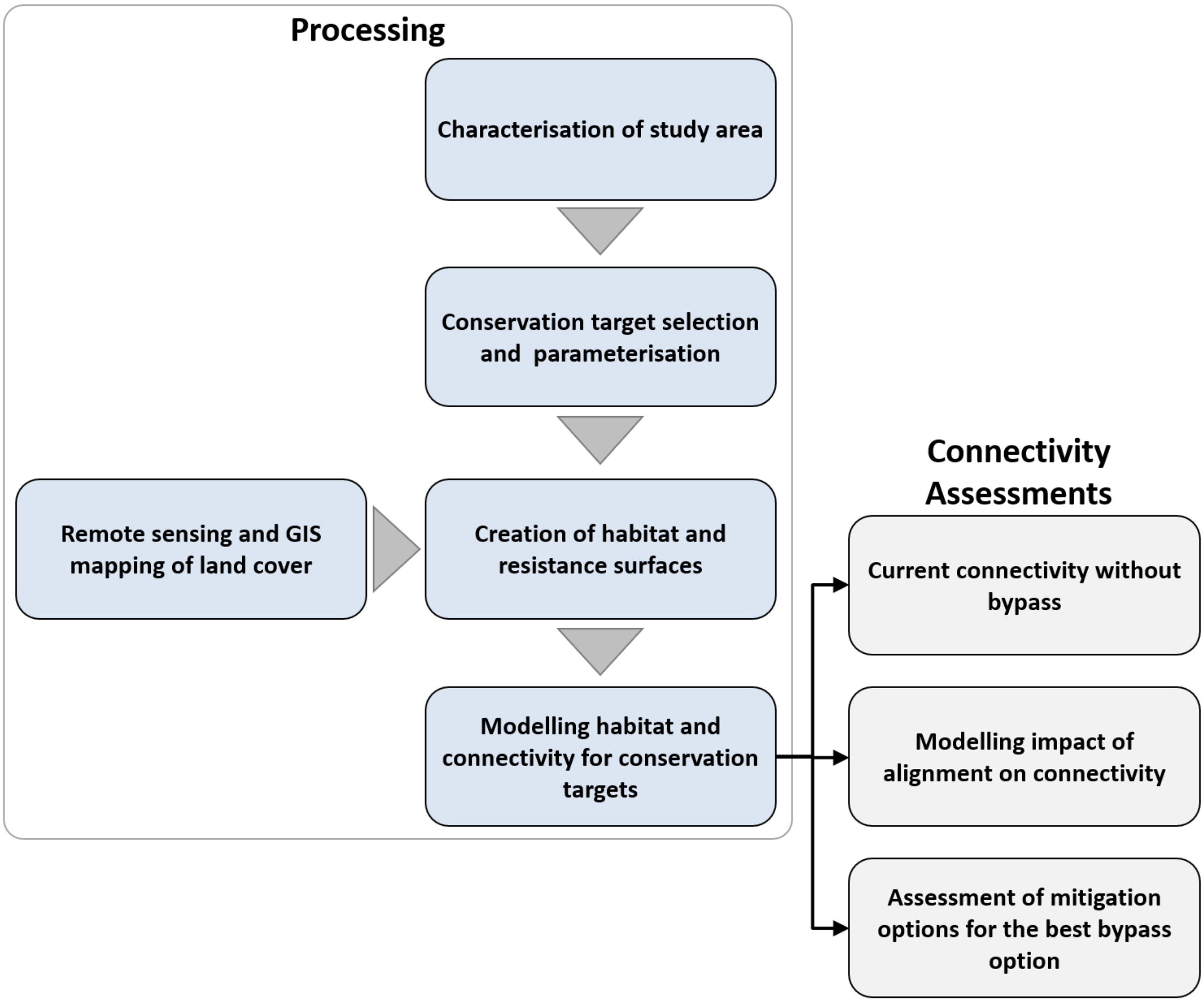

Figure 4.2: Schematic of processing steps to assess the impacts of the Beaufort Bypass on target guilds. *Processing steps on the left were used to produce spatially explicit models of connectivity for the five conservation targets in the study area. The boxes on the right represent the types of analyses conducted to assess the impacts of the different bypass options on movement of species making up the guilds.*

4.2.3 Characterising study area and conservation target selection and parameterisation

We chose five conservation targets to represent a range of dispersal and habitat characteristics in the study area and included species of conservation concern where possible. With the aid of an expert assessment of broad species or fauna groups present in the study area, we determined five 'dispersal guilds' that represented different habitat and dispersal behavioural features such as patch size, habitat structure, dispersal characteristics and resource exploitation (see Table 4.2). The five guilds had very different habitat and/or dispersal characteristics and thus are likely to respond differently to each of the alignments. The selected conservation targets resulted from balancing data availability (i.e., habitat distribution data and data on the parameterisation of dispersal characteristics) with modelling a diverse range of dispersal and habitat characteristics (Table 4.1). Following Lechner et al. (2017) the aim of conservation target selection was to ensure that we included sufficient diversity in the range of conservation targets modelled to represent a wider diversity of species. All the conservation targets apart from woodland birds were single species (Table 4.1). The conservation target species chosen for each guild were: i) Short-beaked Echidna, representing long dispersing (>1000 km) ground dwelling mammals dependant on woody vegetation habitat, ii) Brush-tailed Phascogale, representing long dispersing (>1000 km) arboreal mammals dependant on woody vegetation habitat, iii) Growling grass frog, representing short dispersing (<1000 km) amphibians dwelling in riparian and wet habitats, iv) Golden Sun Moth, representing short dispersing (<1000 km) invertebrates residing in grasslands and grassy woodlands, and v) woodland birds, signifying tree-cover sensitive birds dependant on woody vegetation as habitat. For woodland birds, general woody vegetation parameters were used to represent bird assemblage synonymous with the Victorian Temperate Woodland Bird Community listed under the Flora and Fauna Guarantee Act 1988 (FFG Act) (see Table B 1 for full list).

Table 4.1: Conservation targets and their characteristics, general species or fauna groups they represent and their habitat and dispersal characteristics. *Parameters used for connectivity modelling are also given. Resistance values are given as multipliers of the cost to move through optimal habitat (i.e., a value of 1 = no cost) and a resistance value of 2 means the land cover type is twice as costly or difficult to travel through.*

Parameter		**Woodland birds**	**Short-beaked Echidna** *Tachyglossus aculeatus*	**Brush-tailed Phascogale** *Phascogale tapoatafa*	**Growling Grass Frog** *Litoria raniformis*	**Golden Sun Moth** *Synemon plana*
Habitat group	Broad habitat group	Woody Vegetation	Woody Vegetation	Woody Vegetation	Riparian and Wet habitat	Grassland and Grassy woodland
	Taxonomic group	Tree-cover sensitive birds	Ground-dwelling mammal	Arboreal mammal	Amphibian	Invertebrate
	Dispersal characteristics	Representative of the 'average species' dispersal characteristics (cf.. Lechner and Lefroy 2014).	Long disperser, Woody-vegetation dependent.	Long disperser, Woody-vegetation dependent	Short disperser, Wetland dependant	Short disperser, Grassland dependant,
Movement and patch size	Minimum Habitat Patch Size (ha)	5	50	45	0.000082	0.00046
	Gap-crossing threshold (m)	106	500	750	500	200
	Interpatch-crossing distance threshold (m)	1000	5000	5500	500	200
Resistance (multiplier)	No resistance i.e., pasture, open grasslands	1	1	1	1	1
	Woody vegetation (used for gap-crossing layer)	1	1	1	1	1
	Waterways and water bodies	3	2	3	1	1.5
	Residential	2	10	2	2.5	2
	Rail	2	2	2	1.5	1.5
	Roads	2	1.5	3	1.5	1.5
	Bypass and existing highway	10	Infinite	20	10	10

Source	(Lechner and Lefroy 2014; O'Malley and Lechner 2017)	(O'Malley and Lechner 2017)	(Eco Logical Australia 2012)	(O'Malley and Lechner 2017)	(O'Malley and Lechner 2017)

4.2.4 Pre-processing spatial data based on parameters defined

4.2.4.1 Land cover and habitat mapping

A land cover map was created to identify habitat and characterise movement costs for each conservation target. We used a combination of automated image classification and manual interpretation to derive land cover classes. All spatial layers were created at 2 m pixel size due to computational limitations and limitations around mapping precision. However, this was a relatively high spatial resolution compared to other connectivity models (Lechner et al. 2015c; Lechner et al. 2017) and was enough to model the fine-scaled movement patterns for all conservation targets. The classified land cover map was further refined and supplemented by existing spatial data of relevance (crowd sourced data and mapped infrastructure and environmental datasets). This included information such as vegetation mapping (extant and modelled), existing land use mapping (transport networks, land tenure datasets, building footprints), hydrological datasets for waterways, wetlands, water bodies and associated assets. Linear infrastructure was buffered by its average width before being rasterised. The Western Highway was divided into two parts which we mapped and modelled differently. The portions leading up to the potential bypass design from the west and east were dual carriageways with a large median strip. The sections of the Western Highway sections nearer to town are single carriageway and of a similar width to the other major roads in the study area. Each landcover dataset was processed in ArcMap 10.6.1 and combined into a landcover map (Figure 4.3).

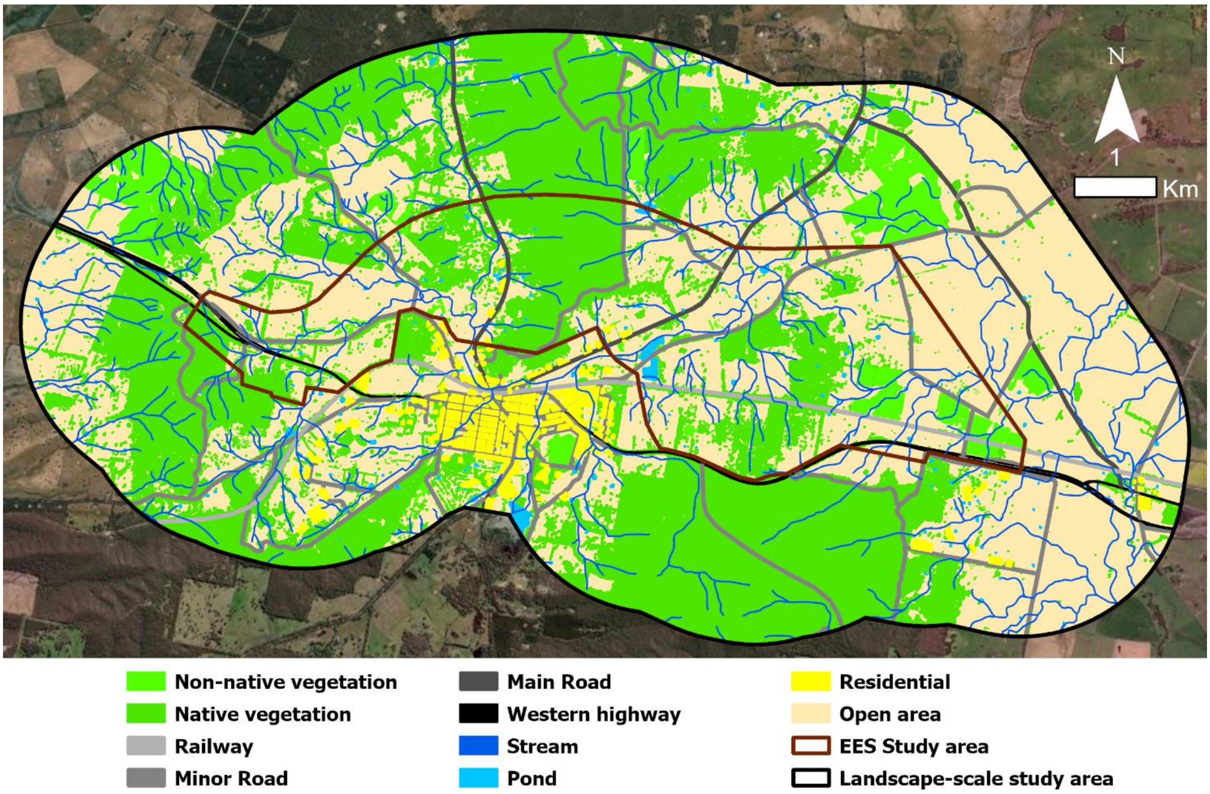

Figure 4.3: Landcover map used for modelling connectivity.

4.2.4.2 Gap crossing layer

The gap-crossing layer characterises distances between connectivity elements (e.g., scattered trees) and patches based on a gap-crossing distance threshold unique to each target species and guild. Any habitats smaller than the minimum patch size act solely as connectivity elements, allowing animals to only use them to move through the landscape; i.e., they are too small to support viable populations. These connectivity elements represent land cover features such as scattered trees or linear roadside vegetation. To model gap-crossing behaviour, all connectivity elements (e.g., scattered trees) and patches were buffered by half of the species-specific gap-crossing distance threshold (Table 4.1). Movement is only allowed within areas where buffers touch or overlap whereas areas outside of the buffered areas are considered as barriers to dispersal and represent areas where distances between connectivity elements and patches are greater than the gap-crossing distance. Thus, dispersal can only take place in areas within the gap crossing distance. For both the Growling Grass Frog and Golden Sun Moth, we did not create a gap-crossing layer as data to provide high resolution mapping of fine-scale elements for these species were unavailable.

4.2.4.3 Characterise land cover resistance

Resistance to dispersal is described by the cost of movement unique to each species and land cover and how it reduces the maximum cumulative distance individuals can travel between patches (interpatch-crossing distance threshold). For each conservation target a unique resistance surface was created by assigning resistance values to each landcover class (Figure 4.3) based on the resistance values in Table 4.1. These resistance values were derived from existing studies which were themselves based on expert-based workshops/interviews and literature sources (Eco Logical Australia 2012; Lechner and Lefroy 2014; Lechner et al. 2017; O'Malley and Lechner 2017). Finally, we combined the gap-crossing layer with the resistance surfaces. The gap-crossing layer has priority over the land cover map, which means that areas which are beyond the gap-crossing threshold are represented as a barrier to movement. The resulting dispersal cost layer is one that acknowledges fine-scale threshold dynamics as it ensures dispersal is impossible where gaps are greater than the gap-crossing distance.

4.2.5 Modelling present connectivity for conservation targets

Each conservation target had unique input spatial layers to be used with the connectivity modelling software. Graphab 2.2 software was used to conduct the least-cost path analysis (Foltête et al. 2012) and Linkage Mapper with Circuitscape (McRae et al. 2008; McRae and Kavanagh 2011) was used to model movement based on current density (McRae et al. 2008).

First, patches were identified based on a minimum patch size for all target species except for the frog and moth, by selecting patches of woody vegetation greater than the patch-size threshold (Table 4.1). All areas characterised as habitat for the frog and moth were included, regardless of their size. Next, least-cost paths were generated to identify the optimal paths between all patches. A least-cost path between two patches will exist if the cumulative cost distance based on the resistance surface is below the interpatch dispersal distance threshold. Additionally, we also identified areas of isolation and fragmentation represented by 'component boundaries'. These were identified by Graphab as the midpoint between two patches where a least-cost path does not occur, thereby delineating the boundary of interconnected habitat. Spatial patterns of these components are useful for characterising fragmentation and barriers to connectivity at the regional scale (Lechner et al. 2015c). The size and shape of these components and the number of patches they contain can describe the levels of fragmentation and pinpoint where barriers to connectivity exist within the broader landscape.

Linkage Mapper was then used to calculate least-cost corridors. A least-cost corridor is a corridor between two patches and has a width limited by the cost-weighted distance threshold defined by the resistance surface and interpatch dispersal distance threshold. Areas beyond the least-cost corridors represent parts of the matrix which are not utilised for movement. Circuitscape was used to characterise areas of high movement probability within these corridors based on random walk patterns between patches using circuit theory (McRae et al. 2008). These corridor areas are represented by high current density values restricted to a particular location between patches. These locations will often overlap with the least-cost path, providing an indication of which parts of the least-cost path have low redundancy. This analysis is important for characterising the redundancy of the least-cost pathways which only represent a single optimal pathway between patches.

Landscape connectivity for each of the conservation targets was analysed by assessing the degree of patch isolation (component boundaries) and the distribution of least-cost paths. In addition, graph metrics were used to rank linkages and patches quantitatively. We calculated one patch-scale metric and four landscape-scale metrics (Table 4.2). The patch-scale metric delta Integral index of connectivity (dIIC) (Pascual-Hortal and Saura 2006; Saura and Pascual-Hortal 2007) was used to represent the importance of a linkage or patch for connecting the landscape by expressing the change in habitat availability caused by the elimination of the focal patch. A higher dIIC value denotes that a linkage or patch is important for connecting habitat in the study area. Landscape-scale metrics provide a single numerical value that characterises how well-connected a landscape is. We calculated the following metrics: Mean size of components, Size of largest components, Number of components, and Integral Index of Connectivity (IIC). While most of the metrics we used have literal interpretations, the IIC (Pascual-Hortal and Saura 2006; Saura and Pascual-Hortal 2007) is the landscape-scale form of dIIC. IIC values increase with greater connectivity from zero to one and is best used as a relative measure to support what could be visually seen with the least-cost path assessment map.

4.2.6 Modelling impact of different bypass routes on connectivity

Impacts of four alternative bypass alignment options on connectivity were modelled. A nominal construction footprint was devised, using the outer limit of the functional road design for each alignment including pavement surfaces, batter slopes, cuttings, drain inverts and other earthworks. This was then buffered by 10 m to represent the area that may be impacted by construction. The construction footprint was used to erase any vegetation within map layers, thus affecting dispersal by increasing the gap crossing distance. The impact of the different bypass alignment options was assessed by comparing the connectivity under the present landcover with the different potential scenarios, based on changes in the number and distribution of least-cost paths, components and dIIC. In addition, we calculated the relative difference in landscape metrics between the present and each potential alignment scenarios. The percentage change in IIC between the present and future scenarios can be used to quantify the impact of each of the alignments on landscape connectivity. IIC incorporates total habitat so design options which remove more habitat will also reduce IIC values even if connectivity remains the same.

4.2.7 Assessment of mitigation options for the best bypass option

After determining the preferred alignment, we modelled the likely benefits of wildlife crossing structures (namely eight canopy rope bridges and a vegetated land bridge) on species dependent of native woody vegetation (i.e., Woodland birds, Short-beaked Echidna and Brush-tailed Phascogale). The locations and their WCS type were selected by based on expert opinion or the composition of the landscape (Clevenger et al. 2002; Cushman et al. 2014; Liu et al. 2018; Wierzchowski et al. 2019) and were identified based on having adequate tree cover near the road and on locations where the functional design permits these types of structures. For example, canopy rope bridges were identified at locations where the bypass was at grade or in a cutting while the vegetated land bridge was proposed for a location where the bypass was in a cutting.

We also calculated the increase in connectivity associated with the addition of each mitigation structure using IIC. We modelled the use of the vegetated land bridge for the Short-beaked Echidna and Woodland birds, and considered rope bridges in addition to the land bridge, for Brush-tailed Phascogales.

4.3 Results

4.3.1 Present connectivity

From the perspective of woodland birds, the landscape is fragmented into two large groups of interconnected patches. This is likely to affect guild viability due to the well-studied negative impacts of isolation (Doerr et al. 2011; Fahrig et al. 2019). Areas of high movement probability are shown as yellows and low movement probability areas as dark blue on Figure 4.4. A critical location for the woodland birds is in the larger component in the northwest of the study area. Within the chain of high dIIC patches and linkages there is a high movement probability corridor among the patches in the north and patches in the southwest. This area is critical for maintaining the northwest component as a single component.

For the Short-beaked Echidna and Brush-tailed Phascogale, habitat patches within the study area are connected within a single component (Figure 4.4b and c) due to their particularly long interpatch dispersal distances relative to the study area. The distribution of dIIC values for patches and linkages for the Short-beaked Echidna showed that one of the most critical linkages is between the central patch in the north and another large patch in the southeast (Figure 4.4b). As for the Brush-tailed Phascogale, one of the most critical linkages is between the central northern patch and another large patch in the southeast of the study area (Figure 4.4c). The Linkage Mapper analysis shows that both species are likely to utilise most of the matrix for dispersal because of their long interpatch dispersal distances and gap-crossing distances, and therefore there are few locations where dispersal is constrained.

The landscape for the Golden Sun Moth and the Growling Grass Frog is highly fragmented, as most patches are either isolated or connected to a small number of patches (Figure 4.4d and e). The distances between most of their habitats within the study are much further than their interpatch dispersal distances thus are unlikely to be connected. The high dIIC areas for these two species are mostly made up of large patches and connecting linkages. The Linkage Mapper analysis suggests that not much of the matrix is utilised for dispersal by both species due to their short interpatch dispersal distance and there are very few locations with high movement probability.

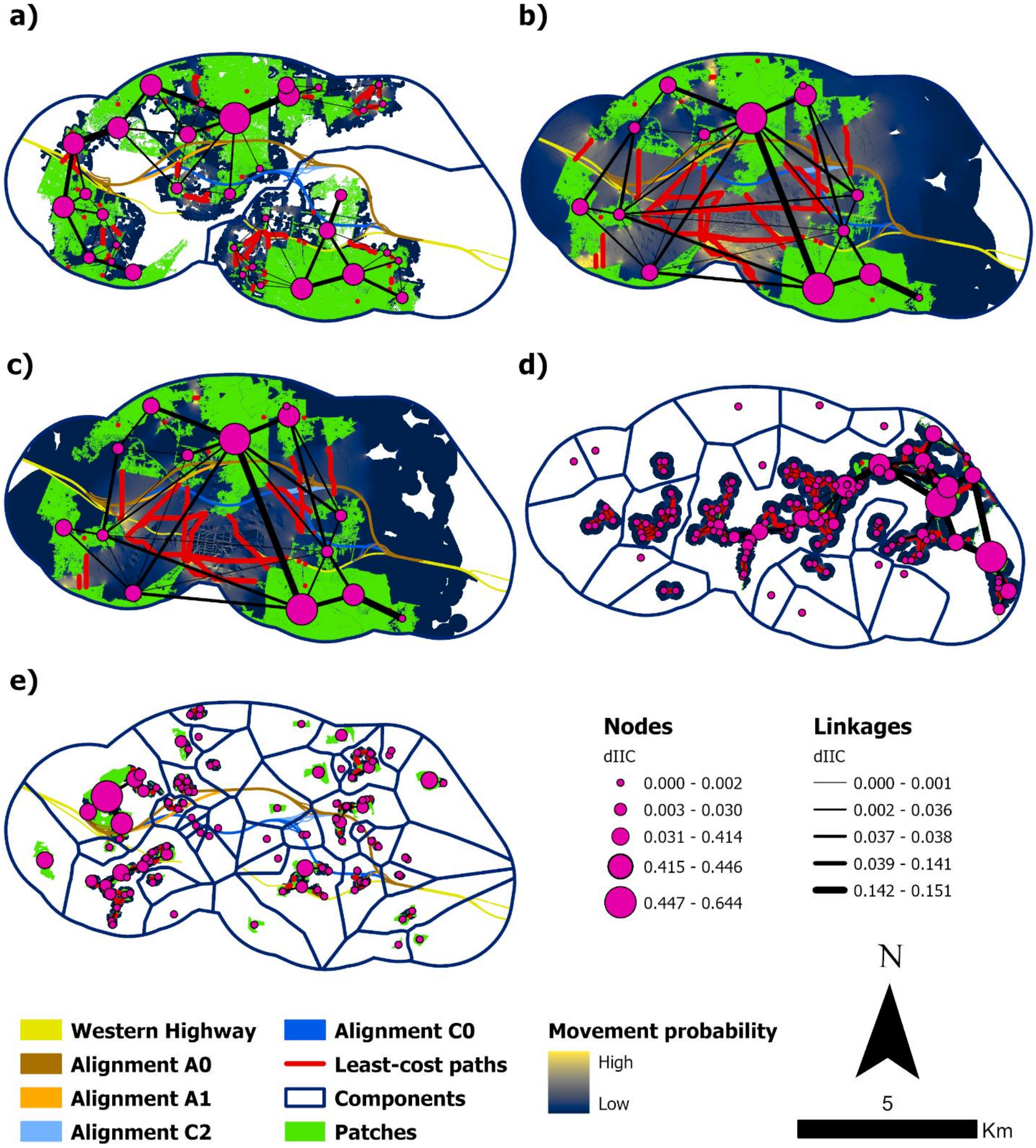

Figure 4.4: Present habitat patches for a) Woodland birds b) Brush-tailed Phascogale c) Short-beaked Echidna d) Golden Sun Moth e) Growling Grass Frog with component boundaries, Least-Cost (LC) paths and delta Integral Index of Connectivity (dIIC) for patches and linkages. *Important linkages and patches are*

denoted by thick lines and large circles respectively. The circles located at the centroid of each patch describe patch-scale graph metric values. Lighter areas represent areas of high movement probability while darker areas represent areas where no movement occurs. The bypass designs are included (but not modelled) to allow for a comparison with present connectivity.

4.3.2 Impacts of bypass alignment options

Each of the conservation targets are impacted differently by each alignment option (Figure 4.5 for Alignment C2, see Figure B 1-3 for effects of all other alignments for each species). The potential impacts of the four bypass alignments on present levels of connectivity for each species is quantified in Table 4.2. Landscape-scale metrics consistently show alignment options rank in the order A0, A1, C0, C2 (least to most impact). Alignment A1 removes the most habitat and reduces connectivity by the largest degree across all target species. Alignments A1 and C0 have moderate impacts but A1 still has quite high impacts on connectivity for 3 of the 5 conservation targets. Alignment C2 has the least impact overall on connectivity and habitat loss, across all target species.

For woodland birds, the group of patches in the north would be most affected as all four bypass designs intersect them. However, none of the bypass alignment options resulted in the creation of new components, indicating that the creation of new patches and changes to resistance did not result in any patches or groups of patches becoming isolated. The decrease in total patch area (i.e., greatest amount of habitat lost) is the greatest with Bypass Design A0, followed by C0 and A1. Habitat patches are least fragmented with bypass design C2 and most fragmented with A0, C0 and A1. Graph metrics related to the characteristics of the components appear to have changed little, primarily because none of the design options result in the creation of new components. According to the IIC analysis as well as visual and quantitative assessments of the design options, bypass design C2 has the smallest relative impact, decreasing IIC value by 2.4% and avoiding majority of the habitat patches.

The Short-beaked Echidna and Brush-tailed Phascogale, both long-distance dispersers, were relatively less impacted overall by each bypass alignment option than other species. Habitat patches for the Short-beaked Echidna were unaffected except for design A0 in which the number of patches was reduced by one patch (Figure A 2).

However, the IIC assessment showed a large decrease in the IIC value of 40% or greater for all bypass designs. This is because the bypass will fragment the landscape into two components for this species. The differences between the alternate bypass alignments on the Brush-tailed Phascogale was mostly driven by the loss of habitat. None of the alignments caused isolation of any patches in the landscape and therefore all patches are found within a single component regardless of the alternative alignments. For both species, the assessment of overall IIC values show that option C2 has the least impact on functional connectivity and habitat loss for these two species.

Impacts of each alignment option for the Golden Sun Moth and Growling Grass Frog were very small. The differences in IIC values between the designs were less than 0.1% for the moth and 0.01% for the frog. For the moth, habitat patches will be least fragmented with alignment C2 followed by A1, A0 and C0. The number of components increased the most with Bypass Designs A0 and C0 with an increase of four components. Surprisingly, the bypass did not result in the isolation and creation of new components for the frog as despite the presence of the bypass, least-cost paths still exist across the bypass as the patches are very close to each other. Overall, design C2 had the least impact quantitatively (in terms of IIC and total area) for both species, but it should be noted that all designs have broadly similar and limited impacts on both, given the present levels of habitat loss and fragmentation.

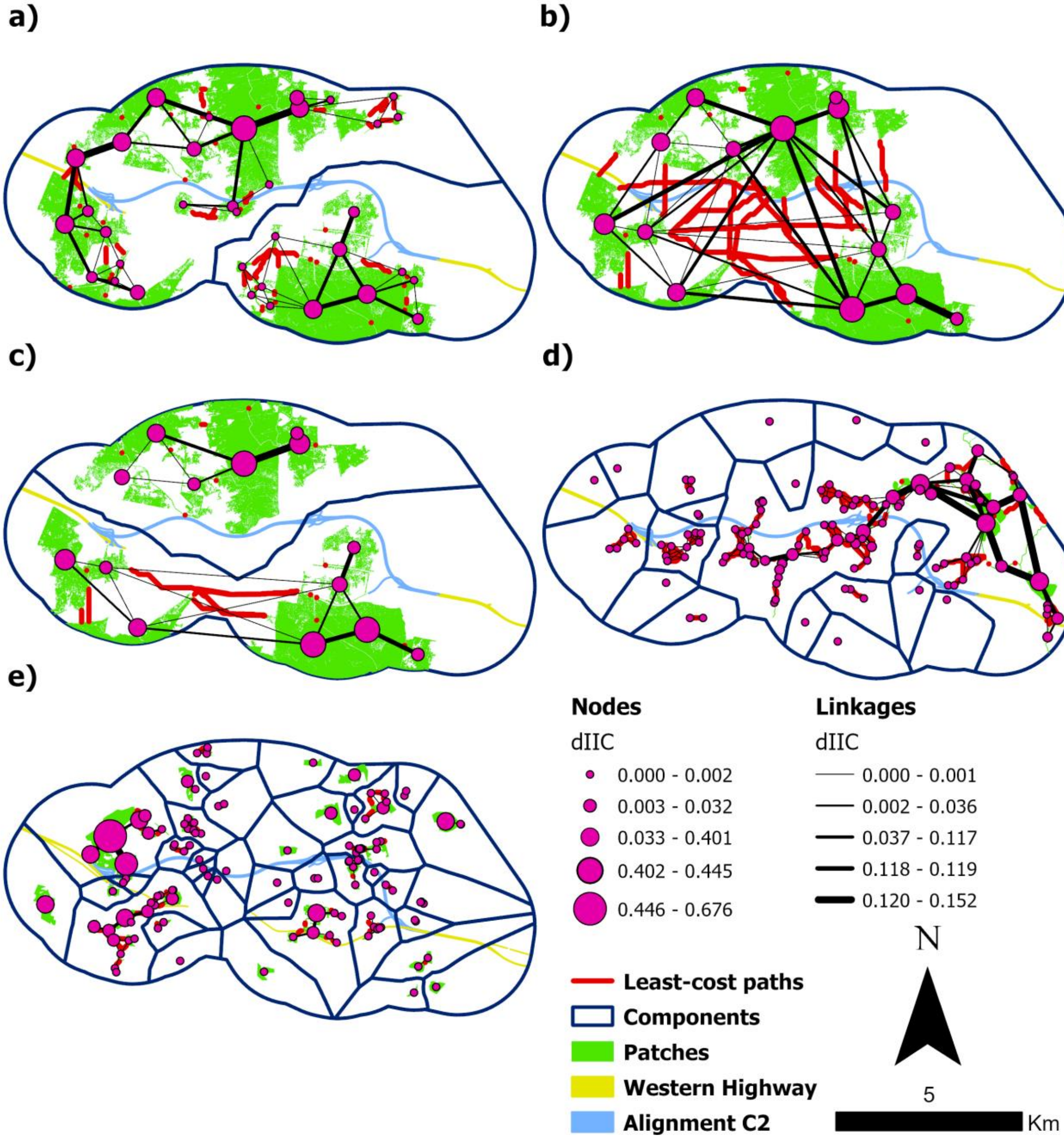

Figure 4.5: Future scenario modelling for Bypass alignment C2 for a) Woodland birds b) Brush-tailed Phascogale c) Shorth-beaked Echidna d) Growling Grass Frog e) Golden Sun Moth. *The figures include the habitat patches with component boundaries and delta Integral Index of Connectivity (dIIC) for patches and linkages. The circles located at the centroid of each patch describe patch-scale graph metric values. Important linkages and patches are denoted by thicker lines and larger circles respectively.*

Table 4.2: Landscape-scale graph metrics and the number of patches and links for all scenarios tested for each species. *IIC and total patch area are correlated, with the impacts colour coded from red to green; where green means least impact and red means greatest impact.*

Graph Metric	Species	Present Scenario	Future Scenario: Design Option A0	Future Scenario: Design Option A1	Future Scenario: Design Option C0	Future Scenario: Design Option C2
Mean size of components (ha)	Birds	1228.5	1216.8	1218.6	1218.6	1223.8
	Brush-tailed Phascogale	2200.1	2121.8	2121.8	2127.2	2179.7
	Short-beaked Echidna	2200.1	1060.9	1061.5	1063.6	1089.9
	Golden Sun Moth	13.7	12.1	12.7	12.1	12.7
	Growling Grass Frog	11.9	10.7	10.7	10.6	11.3
Size of largest component (ha)	Birds	1577.9	1554.7	1558.4	1568.7	1568.7
	Brush-tailed Phascogale	2200.1	2121.8	2121.8	2127.2	2179.7
	Short-beaked Echidna	2200.1	1292.4	1292.3	1072.1	1172.5
	Golden Sun Moth	161.3	148.2	148.1	148.1	148.1
	Growling Grass Frog	203.8	203.1	203.1	201	203
Number of components	Birds	2	2	2	2	2
	Brush-tailed Phascogale	1	1	1	1	1
	Short-beaked Echidna	1	2	2	2	2
	Golden Sun Moth	39	43	41	43	41
	Growling Grass Frog	18	20	20	20	19
IIC	Birds	0.012785	0.012431	0.012049	0.012344	0.012479
	Brush-tailed Phascogale	0.019285	0.0174386	0.0174386	0.019494	0.019279
	Short-beaked Echidna	0.019138	0.010909	0.010919	0.010933	0.011317
	Golden Sun Moth	0.00022892	0.00020258	0.00020249	0.00020271	0.00020262
	Growling Grass Frog	0.00013399	0.00013005	0.00013004	0.00013080	0.00013004
Patches	Birds	34	36	36	36	35
	Brush-tailed Phascogale	14	14	15	13	14
	Short-beaked Echidna	14	14	14	13	14
	Golden Sun Moth	118	125	124	128	123
	Growling Grass Frog	127	125	124	130	124
Links	Birds	61	81	66	62	60
	Brush-tailed Phascogale	31	35	36	31	33
	Short-beaked Echidna	29	22	22	16	19
	Golden Sun Moth	120	128	127	132	126
	Growling Grass Frog	199	184	183	199	188
Total patch area (ha)	Birds	2456.9	2433.6	2437.3	2437.2	2447.6
	Brush-tailed Phascogale	2200.1	2121.8	2172.7	2127.2	2179.7
	Short-beaked Echidna	2200.1	2121.8	2122.9	2127.2	2179.7
	Golden Sun Moth	535.9	520.5	521.8	520.8	522.4
	Growling Grass Frog	215	213.8	214.2	212.4	214.1

4.3.3 Likely effects of mitigation measures

The conservation target that benefitted most from the addition of mitigation measures was the Echidna. Without the land bridge, the bypass would represent a complete barrier to dispersal for this species, and with the land bridge, there was a 71.6% increase in IIC for this species (Figure 4.6b). Connectivity for the Brush-tailed Phascogale increased by 41.9% after all nine wildlife crossing structures were added, with the land bridge contributing the greatest increases (Figure 4.6). The addition of a land bridge increased connectivity for woodland birds by only 1.7% (IIC increased from 0.01248 to 0.01269) (Figure 4.6a). The impact of the overpass was relatively small as the two patches on opposite sides of the bypass in Figure 4.6a were partially connected even without the addition of the land bridge (see Figure 4.4a). The addition of these mitigation measures to design option C2 generated new least-cost paths for each species.

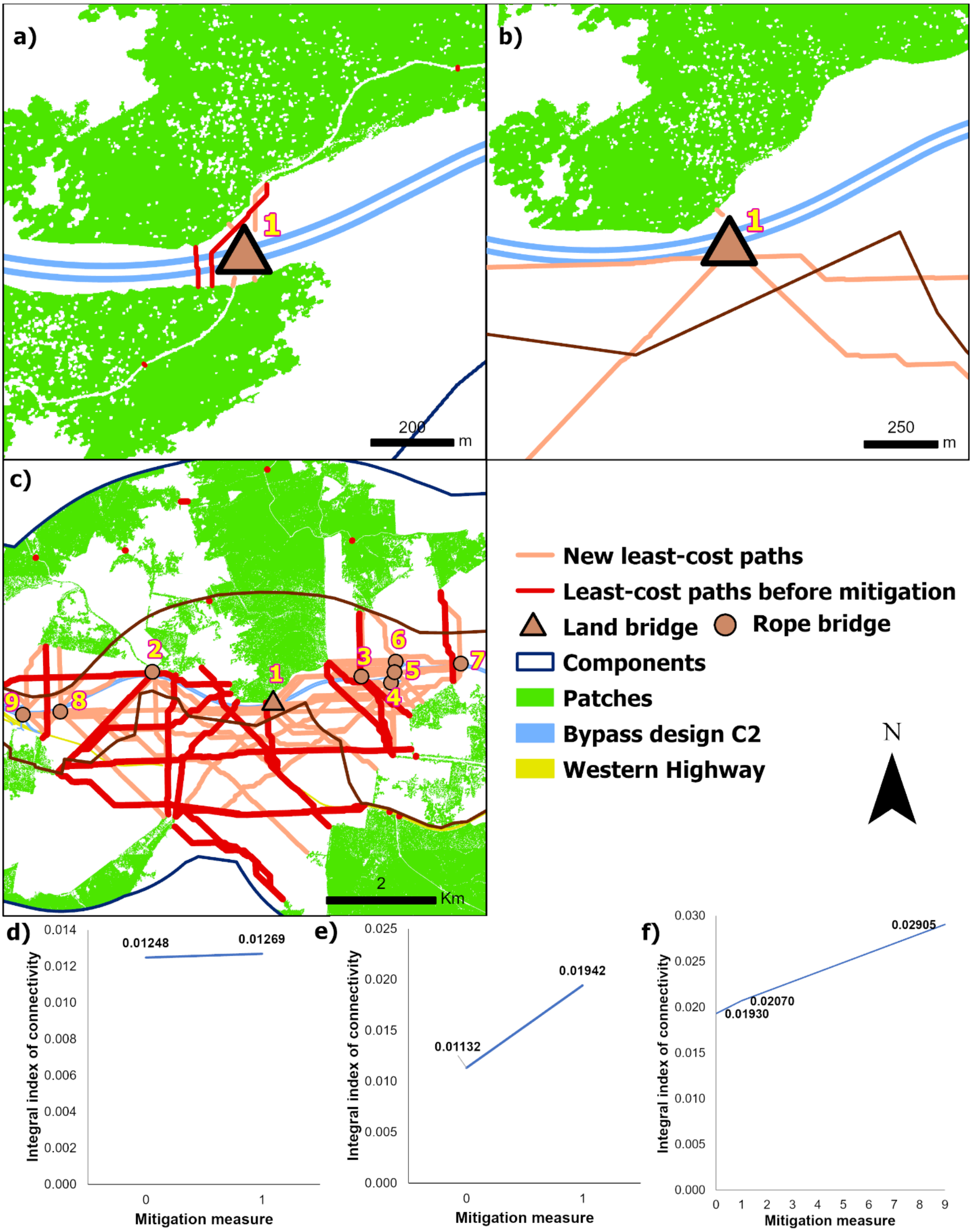

Figure 4.6: New least-cost paths for a) Woodland birds b) Short-beaked Echidna c) Brush-tailed Phascogale along Bypass alignment option C2 after the addition of their respective mitigation measures. *The land bridge is denoted by the Number 1 while 2-9 represent rope bridges.*

4.4 Discussion

There is an increasing amount of empirical research using quantitative techniques to support EIAs (Bacaro et al. 2011; Chiarucci et al. 2011; González Del Campo and Gazzola 2020) but few examples exist of the application of these techniques to understand impacts on ecological connectivity. We addressed this gap by developing and applying a connectivity modelling approach to a real-world EIA, in order to better assess and mitigate impacts of transport infrastructure. We demonstrated how to use data, landscape models, and expert knowledge at the early planning and design stages of a highway project via a multi-scenario and multi-target approach to better understand how connectivity is impacted by different alignment options. The connectivity modelling also provided spatially explicit projections of how each target would respond to the alternate bypass alignment and mitigation options.

The scenario analysis showed that each conservation target responded differently to alternative bypass option scenarios. Simulations indicated a high movement probability corridor between the patches in the north and patches in the southwest for the woodland bird group, as shown by the chain of high dIIC patches and linkages; this corridor is critical for maintaining the northwest component as a single component. Both the Short-beaked Echidna and Brush-tailed Phascogale are likely to utilise most of the matrix for dispersal because of their high dispersal ability, and therefore there are few locations where dispersal is constrained before the construction of the bypass. However, for the Short-beaked Echidna the analysis highlighted that connectivity exists within two large components and this configuration does not change with alternate bypass designs. For short dispersers such as the Golden Sun Moth and Growling Grass Frog, the majority of the landscape matrix is utilised for dispersal due to the short interpatch dispersal distance of these species, with very few locations having high movement probability. Our analysis showed that the landscape was generally already highly fragmented for short distance dispersers (as shown in another Australian multispecies study (see Lechner et al. 2017) and the alignment impacts are likely to be greatest for larger mammals. All alternative alignments had very little impact on connectivity for the moth and the frog due to the already fragmented state of their habitat and their low dispersal ability.

By modelling different scenarios we were able to identify that bypass alignment C2 head the least impact on the target guild and species. This route had the least impact on targets dependent on woody vegetation. For example, the IIC analysis of the bird guild (woodland birds) showed a decrease of only 2.4%, the lowest amongst the alignment options. A visual and quantitative assessment of alignment C2 also showed it avoided majority of the habitat patches for this species. For the Brush-tailed Phascogale, none of the bypass designs caused isolation of any patches in the landscape and therefore all patches are found within a single component regardless of the alternative designs. Conversely, the landscape was divided north to south for the Short-beaked Echidna no matter which alignment was used. This division results from the fact that the models suggested that the bypass represents a total barrier for this species. Nevertheless, assessment of overall IIC values still showed that option C2 had the least impact on connectivity and habitat loss for this species.

This study successfully addressed some of the common challenges of applying spatially explicit ecological assessments in EIAs. Our literature search (Table A 1) indicated that it represented the first work to include both a scenario assessment and evaluation of mitigation options for a road infrastructure development. The only other study that assessed different alignments as well as the likely effectiveness of mitigation focussed on powerlines (Dalloz et al. 2017). Our study showed that the impacts, in terms of total loss of habitat area associated with clearance for constructing the road were relatively small (see Table 4.2), but the impacts on connectivity across the study area were far reaching for certain species.

Our approach demonstrated the importance of modelling fine-scale dispersal patterns (i.e., gap crossing distances), modelling multiple species and the application of quantitative metrics in order to sufficiently understand impacts and mitigation. On a broader scale, calculating landscape metrics provided direct measurements of cumulative effects across the landscape (e.g., Fahrig and Rytwinski 2009; Olsson 2009; Jackson and Fahrig 2011) from the road alignment which allowed for a quantitative comparison of different scenarios. Modelling the four species indicated that those with short dispersal distances and found in discrete habitat patches (i.e. Golden Sun Moth and Growling Grass Frog) perceive the landscape as highly fragmented, while relatively longer-distance dispersers (i.e. Short-beaked Echidna and Brush-tailed Phascogale) are able to move through most of the matrix due to the presence of scattered trees and perceive the landscape as relatively unfragmented, but were the most impacted by the road construction (Connelly 2011; Duinker et al. 2012; Lechner et al. 2017). Importantly, the modelling of gap-crossing behaviours allowed for fine-scaled land cover features such as scattered trees or linear roadside vegetation to be incorporated in our modelling. Chapter 3 showed that connectivity analyses that exclude fine-scaled connectivity elements risks overestimating connectivity within in similar kinds of fragmented agricultural landscapes.

This work provides a practical demonstration of connectivity modelling for a real-world EIA. Firstly, by providing a quantitative and spatially explicit assessment of bypass alignments, project proponents and regulators were able to evaluate the likely consequences of different courses of action. Such scenario-based approaches are amongst the most preferred methods for the creation of alternative options to support impact assessment and decision-making (see Duinker and Greig 2007). Second, by predicting the likely effects of mitigation measures for the least impactful alignment option, we are able provide recommendations to further limit its impact on connectivity for all of our conservation targets. If the location of wildlife crossings are considered before construction (Lesbarrères and Fahrig 2012), they can be more easily incorporated into detailed designs of projects than when trying to retrofit into finalised designs or completed projects. Such mitigation measures are also likely to mitigate impacts for other species in the region with similar dispersal characteristics.

4.5 Conclusion

Our study carried out an appraisal of reasonable alternatives to a proposed highway bypass for the town of Beaufort and took into account the extent to which viable measures might help to avoid, minimise or mitigate adverse effects for the least-impactful alignment option. The quantitative assessment and the interpretation of the least-cost paths and component boundaries presented in this study provide strong support for alignment option C2 as being the least impactful alignment option. Alignment C2 had the least impact on connectivity for woody-dependent species, while the differences between each alignment option were negligible for short-distance dispersers and dispersers with low habitat requirements as the majority of impacts for these species are likely to be from total habitat lost. The modelling demonstrated that different species likely perceive habitat fragmentation in very different ways, highlighting that appropriate spatial scales are needed for such technical assessments. Species which perceive the highway as a barrier will be highly impacted by a road development as it will isolate patches from each other and mitigation measures which restore connectivity are critical for maintaining connectivity for species for such species. The mitigation modelling provided support for an overpass as being important for connecting patches to the north and south for woodland dependent species in our study area. Linear infrastructures such as roads are among the main threats to natural habitat worldwide, yet the application of quantitative connectivity modelling methods is rare. Mainstreaming ecological connectivity modelling with scenario analyses in EIAs has the potential to mitigate these impacts globally.

Chapter 5 Synthesis

5.1 Summary

This thesis investigated the gap in knowledge of ecological connectivity in the context of regional connectivity planning and infrastructure impact assessments. The idea of landscape ecology and application of ecological modelling has been recognised for its potential in supporting land use planning and assessing impacts of infrastructure development. However, research that has effectively incorporated truly fine-scale elements in regional planning is lacking, and ecological modelling in linear infrastructure EIAs is underrepresented in the literature (Chapter 2). To address these research gaps, this thesis characterised connectivity for fragmented landscapes impacted by agriculture (Chapter 3) and infrastructure development (chapter 4) and demonstrated the potential for ecological modelling assessments to support planning in fragmented landscapes. It is argued in Chapter 4 that greater ecological consideration is needed within infrastructure impact assessments globally, and that this can be addressed by adopting stronger landscape ecology-based perspectives.

This final chapter discusses the contributions of this thesis and potential challenges in integrating ecological connectivity modelling tools to support land use planning and management on the regional scale, and the assessment of infrastructure impacts to biodiversity by characterising overall connectivity for fragmented landscapes. Section 5.2 provides answers to the research questions that were set out in Chapter 1. Section 5.3 outlines the challenges to integrate ecological connectivity modelling in land management planning and impact assessment that have become evident as a result of the research conducted for this thesis and sets out some next steps that will move the field forward. Section 5.4 concludes this chapter with some final remarks on this thesis, particularly related to the hope that the field of ecological connectivity field can be more formally integrated with EIAs to improve their ability to assess ecological impacts.

5.2 Research Questions

5.2.1 To what extent have academic literature considered ecological connectivity in the context of infrastructure impact assessments?

The systematic review in Chapter 2 highlighted how so little published ecological connectivity modelling work has been done within infrastructure impact assessments (specifically linear infrastructure) when our work has empirically shown that it is highly practical in assessing wildlife impacts on a large scale. Measuring functional connectivity across the landscape has increasingly become a routine objective of researchers and policy makers to retrospectively assess habitat fragmentation of infrastructure projects to inform conservation priority efforts as remote sensing tools become more widely available and affordable (Rudnick et al. 2012). Multiple scholars have advocated for the explicit inclusion of landscape connectivity in the EIA process (Soderman 2005; Karlson et al. 2014; Berges et al. 2020; Patterson et al.). It has been observed as early as 1997, that habitat fragmentation was rarely considered in impact assessments as a potential impact; habitat fragmentation was mentioned in only 4% of UK Environmental Impact Statements even though many of the projects could potentially introduce barriers to wildlife movement and fragmenting habitat (Thompson et al., 1997). However, the inclusion of predictive models in EIAs to predict future impacts before construction begins are still seldom implemented.

In recent times, the gap between knowledge development and knowledge application in landscape connectivity, has been closing. There is also an increase of instruments for multidisciplinary landscape investigations (Opdam et al., 2002). Yet, the absence of landscape connectivity applications is evident in EIAs. Patterson et al. (2022) surveyed 134 EIA actors and stakeholders regarding their perspectives and experiences with connectivity analysis in the context of EA and reported that 54.1% believe that EIA legislation should be modified to better reflect connectivity. They noted that connectivity was typically limited to the project scale and that the methods used to measure and evaluate connectivity were mostly qualitative. Current literature also fails to effectively delimit the scope of EIAs to consider spill-over effects (Fischer 2006; Albert et al. 2017), and other connectivity aspects. Numerous authors have argued that it is neither conceptually nor practically sound to limit the scale of assessment to the project level (Scolozzi and Geneletti, 2012; Bergsten and Zetterberg, 2013), as impacts on connectivity at this scale are frequently deemed insignificant and consequently ignored.

5.2.2 How does fine-scale connectivity elements influence connectivity patterns within fragmented landscapes, and support connectivity on a regional scale?

Chapter 3 demonstrated that landscapes fragmented by agricultural areas can include numerous scattered trees spaced below the gap-crossing distance threshold. The sensitivity analysis (Figure 3.6) suggested that faunal species with an interpatch dispersal distance threshold of 50 to 100 m or fewer are most susceptible to fragmentation in the study area. As many species fear being exposed in the open matrix, the existence of dispersed trees and the distance between them can affect the overall connectivity of fragmented landscapes.

Our study explicitly incorporates the ecology of fine-scale connectivity by mapping vegetation at a high spatial resolution by modelling a gap-crossing distance, building on prior research (Lechner and Lefroy 2014; Lechner et al. 2015b, c) that addressed the importance of scattered trees. This method is particularly significant in landscapes dominated by pastures, which are typically very open with scattered trees. Models that exclude scattered trees run the danger of misrepresenting connectivity patterns in agricultural environments such as the Karuah-Myall catchments in Chapter 3. In addition to distributed trees, small and linear patches also contribute to connectivity within the study area, as evidenced by the large number of small patches with low CC values in the default scenario (Figure 3.5a). The results of our study are important for informing the management of scattered trees by identifying priority sites for tree conservation and regeneration initiatives. Using a gap crossing threshold in modelling efficiently separates trees that contribute to overall connectivity from those that do not.

5.2.3 How can spatially explicit ecological models be utilised to support linear infrastructure impact assessments?

Chapter 4 highlighted how to leverage data, landscape models, and expert knowledge in the early planning and design stages of a highway project to better understand how connectivity is affected by different alignment alternatives. Using a multi-scenario and multi-target approach, the modelling of connectivity offered spatially precise projections of how each target will react to the alternative bypass alignments and mitigation measures. The scenario study revealed that each conservation target reacted differently to various bypass alignment scenarios. Long distance dispersers, such as the Short-Beaked Echidna and Brush-Tailed Phascogale, are likely to utilise the majority of the matrix for dispersal, therefore there are few locations where dispersal is hindered prior to the building of the bypass. Due of the short interpatch dispersal distance of species like the Golden Sun Moth and Growling Grass Frog, the bulk of the landscape matrix is utilised for dispersal by these species, with very few places having a high movement probability. By running several scenarios, we were able to determine that the bypass alignment C2 would have the least effect on the target guild and species.

This work was successful in addressing a number of the common difficulties associated with implementing spatially explicit ecological assessments in EIAs. According to our literature review (Table A 1), this is the first study to integrate both a scenario assessment and evaluation of mitigation alternatives for the development of a road infrastructure. Our investigation revealed that the total habitat area lost as a result of road construction clearing was rather minimal (see Table 4.2), but the impacts on connectivity across the study area were significant for specific species. Scenario-based approaches are among the most favoured for generating alternative options to aid in impact assessment and decision-making (see Duinker and Greig 2007). Subsequently, estimating the probable consequences of mitigation measures for the alignment option with the least impact, allowed for recommendations for further minimising impacts on connectivity. These mitigating techniques are also likely to reduce the negative effects on other species with comparable dispersal characteristics in the region. The study of the Beaufort Bypass established a transparent, quantitative, and repeatable method for assessing consequences and evaluating the anticipated success of various mitigation strategies.

5.3 Limitations and potential aspects of further research

This thesis provided transparent, quantitative and repeatable approaches to assessing impacts of linear infrastructure developments and evaluating the likely effectiveness of different mitigation measures. Nevertheless, although challenging, further improvements could be made. For example, future studies should recognise that spatial data accuracy, connectivity model type, target species and community and ecological parameterisation will all potentially affect the outcome of the modelling (Lechner et al. 2012; Lechner and Rhodes 2016). In chapter 4, these factors are likely to be addressed implicitly through modelling a range of conservation targets with a diverse range of ecological characteristics, but it would be advantageous to model them explicitly in order to evaluate the utility of results. Another key source of uncertainty is that both approaches in Chapter 3 and 4 modelled species' dispersal patterns that relied on expert-based input. While field assessments could be used to better understand dispersal, obtaining the necessary data is a much greater task, especially for multiple species (Compton et al. 2007; Rudnick et al. 2012). Addressing both types of uncertainty are likely impossible in the timeline and context of an EIA due to the time constraints. Thus, it is recommended that following construction, ongoing monitoring is undertaken and used to inform and adaptive management. Roads have a range of other negative effects on wildlife, ranging from direct impacts such as road mortality to long-term impacts on genetic diversity, and where possible post-project monitoring should include assessment of such affects. This way a broader perspective on the impacts of linear infrastructure can be developed and used to inform future modelling work.

5.4 Final remarks

This thesis modelled connectivity for two fragmented landscapes – a forest prevailing landscape fragmented by a matrix dominated by pasture agriculture, and one impacted by a highway bypass construction. Our approach allowed for realistic fine-scale movement patterns to be characterised from an ecological perspective, as well as the appraisal of reasonable alternatives for an infrastructure development; we took into account the extent to which viable measures might help to avoid, minimise or mitigate adverse effects of an infrastructure development, to help land managers identify important conservation values which are often ignored.

The quantitative assessments and the interpretations of the least-cost paths and component boundaries presented in Chapters 3 and 4 provide strong support for incorporating relevant knowledge about the effects of land use changes on the fauna in an area, identifying dispersal networks then singling out suitable and accessible habitat using ecological models. The modelling work done in this thesis demonstrated that different species likely perceive habitat fragmentation in very different ways, highlighting that appropriate spatial scales are needed for such technical assessments. Linear infrastructures such as roads are among the main threats to natural habitat worldwide, yet the application of quantitative connectivity modelling methods is rare. Mainstreaming ecological connectivity modelling with scenario analyses in regional scale conservation planning and impact assessments have the potential to mitigate these impacts globally.

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Appendix A – Chapter 2

Table A 1: Studies that assessed the impact of proposed and existing linear infrastructure projects on wildlife connectivity.

Infrastructure type	Target taxon	Multi-species	Total study area, km²	Impact assessment approach	Type of assessment	Alternative designs/scenario based	Mitigation measures proposed	Source
Rail	Amphibian	No	4600	Least-cost path, graph theory	Retrospective	No	Yes	(Clauzel 2017)
Rail	Amphibian	No	4600	Least-cost path, graph theory	Prospective	No	No	(Clauzel et al. 2013)
Rail	Amphibians, Insects, broad leaved forest dwelling invertebrates	Yes	120	Least-cost path, graph theory	Prospective	Yes	No	(Karlson et al. 2016)
Road	Mammal	No	363000	Circuit theory	Prospective	No	No	(Fullman et al. 2021)
Road	Amphibians, Birds, Mammals, Reptiles, Insects	Yes	2500	Least-cost path, graph theory	Prospective	No	No	(Bourgeois and Sahraoui 2020)
Road	Insect	No	Unspecified	Graph theory	Prospective	No	No	(Vasas et al. 2009)
Power line	General representative species	Yes	Unspecified	Habitat availability and landscape metrics	Prospective	Yes	No	(Dalloz et al. 2017)
Rail	N/A	No	Unspecified	Connectivity indices	Prospective	Yes	No	(Ortega et al. 2016)
Road	Amphibians	No	Unspecified	Individual-based modelling	Prospective	No	No	(Pontoppidan and Nachman 2013)
Road	Mammal	No	30000	Integrated Landscape Ecological Approach	Prospective	Yes	No	(Fernandes 2000)
Urban connectivity including road network	General representative species	No	6956	Least-cost path analysis, Graph theory	Prospective	Yes	No	(Hou et al 2022)
Rail	N/A	No	3200	Connectivity indices	Both	Yes	No	Mallarach and Marull 2006
Road and Rail	Insect, Amphibian, Reptile, Mammal	Yes	3200	Least cost path analysis, Graph theory,	Prospective	No	No	(Babi Almenar et al 2019)

				Circuit theory				
Road and Rail	Birds, Amphibian, Reptile, Mammal	Yes	578	Least cost path analysis, Graph theory	Prospective	Yes	No	(Sahraoui et al 2021)
Rail	N/A	No	360000	Connectivity indices	Both	Yes	No	Martin et al 2021
Non-specific linear infrastructure	N/A	No	Unspecified	Patch indices	Prospective	Yes	No	Geneletti 2004
Road	Mammal	No	7521	Least cost path analysis, Graph theory	Retrospective	Yes	Yes	Saura and Gurrutxaga 2013
Road	Amphibian	No	Unspecified	Multicriteria analysis	Prospective	No	Yes	Nielsen et al 2012
Road	Mammal	Yes	1240	Least cost path analysis, Graph theory	Prospective	No	Yes	Tarabon et al 2019
Road	Mammal	No	186000	Landscape metrics	Retrospective	Yes	No	Westekemper et al 2021
Road	N/A	No	Unspecified	Fuzzy analytic hierarchy process	Prospective	Yes	No	Cerreta et al 2021
Road	Different dispersal distances	Yes	4000	Connectivity indices	Retrospective	Yes	No	Fu et al 2020
Urban connectivity including road and rail network	Amphibian, Mammal, Reptile, Bird	Yes	2025	Least cost path analysis, Graph theory	Retrospective	No	No	Sahraoui et al 2017
Road	Mammal, Bird	Yes	1600	Least cost path analysis	Retrospective	No	No	Liu et al 2014
Road	General representative species	Yes	26640	Graph theory, Circuit theory	Retrospective	Yes	No	Wei et al 2022
Urban connectivity including road and rail network	Mammal, Bird	Yes	6582	Least cost path analysis, Graph theory	Retrospective	Yes	No	Huang et al 2018
Road	Mammal	No	1000	Least cost path analysis, Graph theory	Retrospective	No	Yes	Loro et al 2015

Road	Mammal, reptile, bird, amphibian	Yes	95000	Least cost path analysis, Graph theory	Retrospective	No	Yes	Tarabon et al 2022
Power line	N/A	No	Unspecified	Least cost path analysis, Graph theory	Prospective	Yes	Yes	Biasotto et al 2022

Appendix B – Chapter 4

Table B 1: Victorian temperate woodland bird community species list.

	Common Name	Scientific Name
1	Painted Button-quail	*Turnix varia*
2	Bush Stone-curlew	*Burhinus grallarius*
3	Red-tailed Black-Cockatoo	*Calyptorhynchis banksii graptogyne*
4	Little Lorikeet	*Glossopsitta pusilla*
5	Superb Parrot	*Polytelis swainsonii*
6	Swift Parrot	*Lathamus discolor*
7	Turquoise Parrot	*Neophema pulchella*
8	Barking Owl	*Ninox connivens*
9	Brown Treecreeper	*Climacteris picumnus victoriae*
10	Speckled Warbler	*Chthonicola sagittata*
11	Western Gerygone	*Gerygone fusca*
12	Regent Honeyeater	*Xanthomyza phrygia*
13	Yellow-tufted Honeyeater	*Lichenostomus melanops meltoni*
14	Fuscous Honeyeater	*Lichenostomus fuscus*
15	Black-chinned Honeyeater	*Melithreptus gularis*
16	Brown-headed Honeyeater	*Melithreptus brevirostris*
17	Painted Honeyeater	*Grantiella picta*
18	Jacky Winter	*Microeca fascinans*
19	Red-capped Robin	*Petroica goodenovii*
20	Hooded Robin	*Melanodryas cucullata*
21	Grey-crowned Babbler	*Pomatostomus temporalis*
22	Ground Cuckoo-shrike	*Coracina maxima*
23	Apostlebird	*Struthidea cinerea*
24	Diamond Firetail	*Stagonopleura guttata*

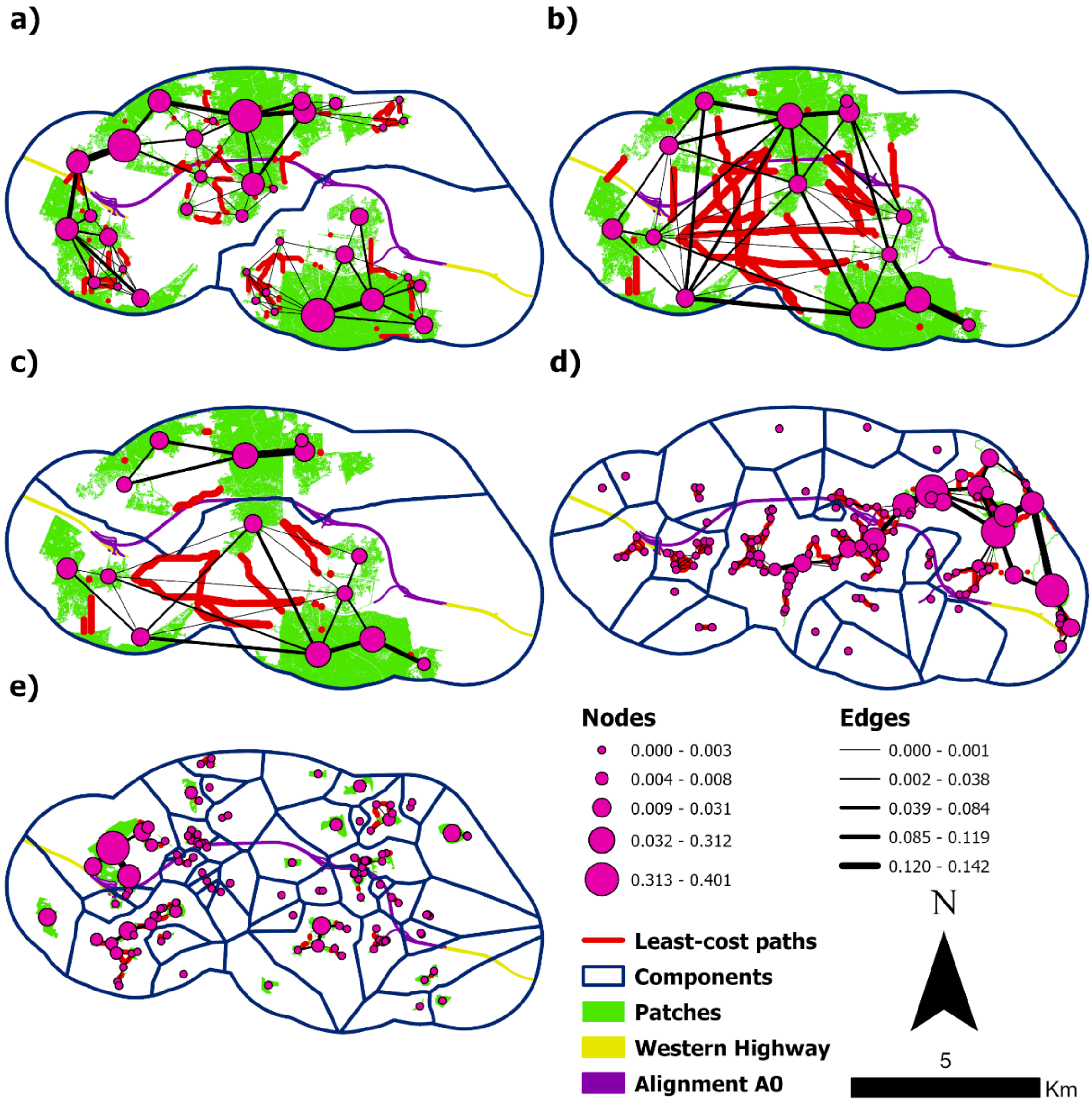

Figure B 1: Future scenario modelling for Bypass alignment A0 for a) Woodland birds b) Brush-tailed Phascogale c) Shorth-beaked Echidna d) Growling Grass Frog e) Golden Sun Moth. The figures include the habitat patches with component boundaries and delta Integral Index of Connectivity (dIIC) for patches and linkages. The circles located at the

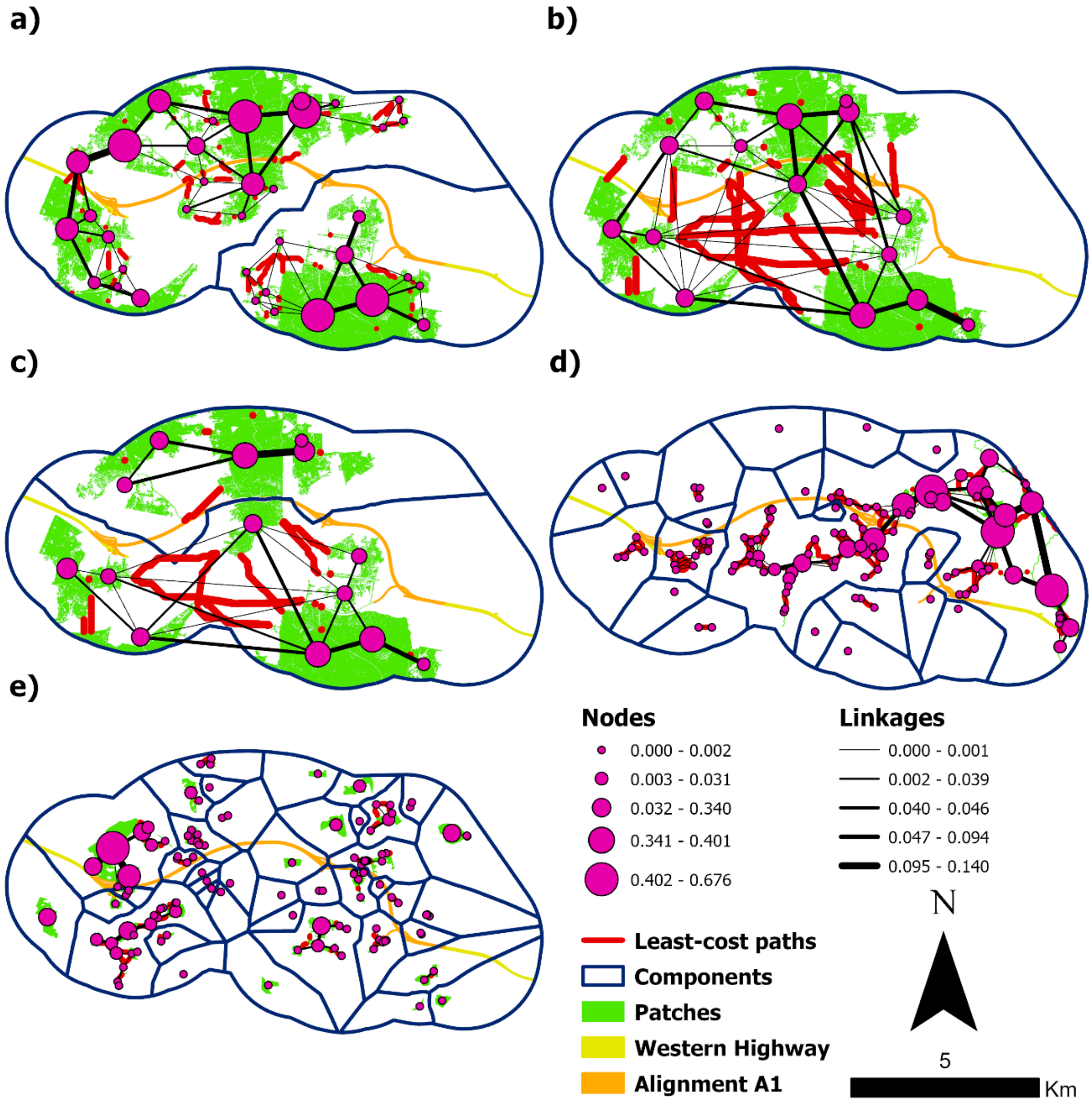

Figure B 2: Future scenario modelling for Bypass alignment A1 for a) Woodland birds b) Brush-tailed Phascogale c) Shorth-beaked Echidna d) Growling Grass Frog e) Golden Sun Moth. The figures include the habitat patches with component boundaries and delta Integral Index of Connectivity (dIIC) for patches and linkages. The circles located at the centroid of each patch describe patch-scale graph metric values. Important linkages and patches are denoted by thicker lines and larger circles respectively.

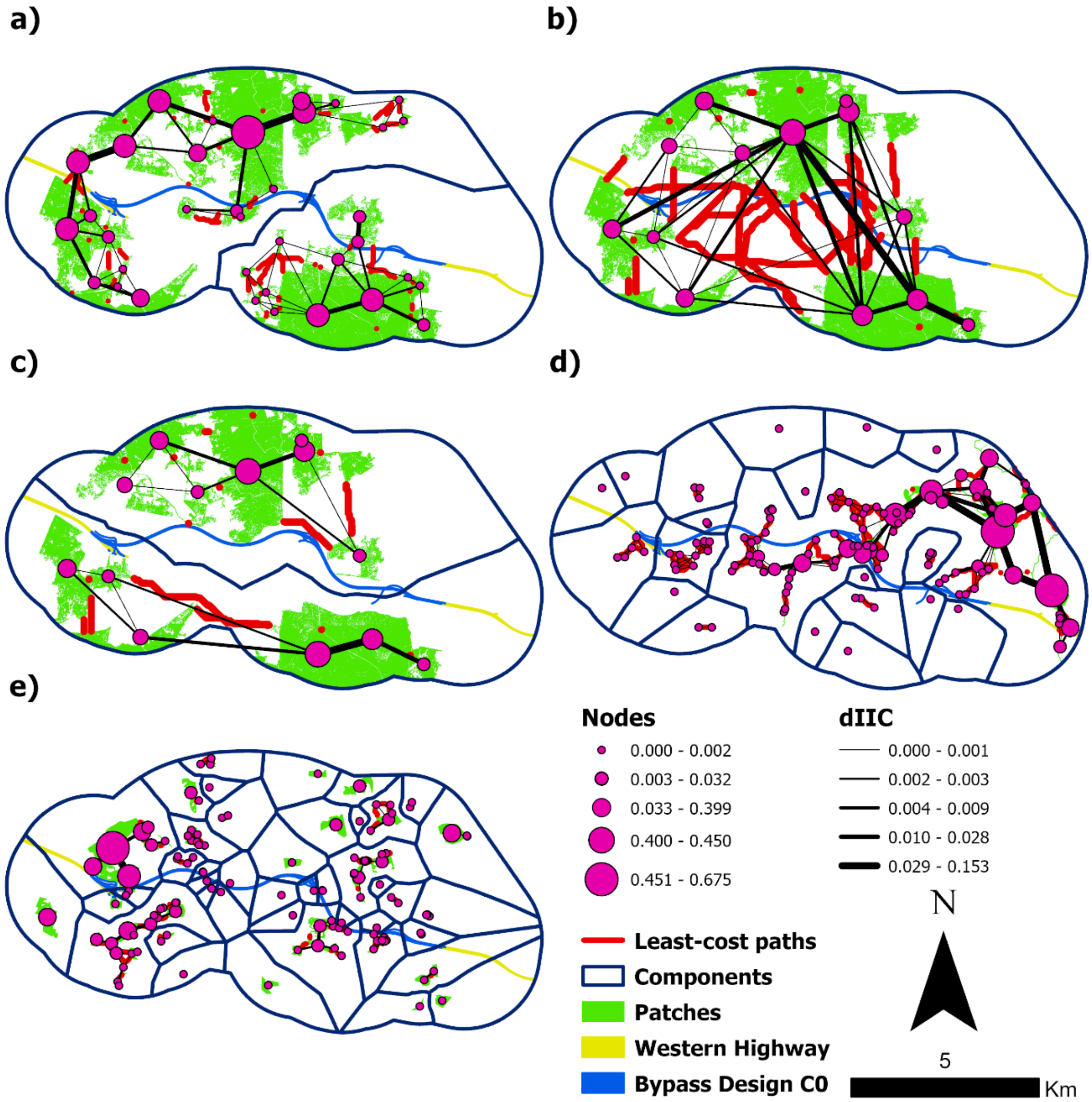

Figure B 3: Future scenario modelling for Bypass alignment C0 for a) Woodland birds b) Brush-tailed Phascogale c) Short-beaked Echidna d) Growling Grass Frog e) Golden Sun Moth. The figures include the habitat patches with component boundaries and delta Integral Index of Connectivity (dIIC) for patches and linkages. The circles located at the centroid of each patch describe patch-scale graph metric values. Important linkages and patches are denoted by thicker lines and larger circles respectively.