Murdoch University

THESIS

Prioritising Rehabilitation for Ecological Connectivity: A multi-scale, multi-species framework to optimise green infrastructure planning in a global biodiversity hotspot

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BSc. Environmental Science (min. Sustainable Development)

This thesis is presented for the degree of
Masters Research with Training in urban landscape ecology at Murdoch University

School of Environmental and Conservation Sciences

2026

This research was supported by an Australian Government Research Training Program (RTP) Scholarship (doi.org/10.82133/C42F-K220).

DECLARATION

I, Taylor D. Page, verify that in submitting this thesis:

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ChatGPT	Structure and readability advice. Statistical and coding advice.

ATTRIBUTION STATEMENT

In accordance with the Murdoch University Graduate Degrees Regulations, it is acknowledged that this thesis represents the work of the Candidate with contributions from their supervisors and, where indicated, collaborators. The Candidate is the majority contributor to this thesis with no less than 75% of the total work attributed to their efforts.

Candidate	Principal Supervisor

ACKNOWLEDGEMENTS

The past two years have been an incredible period of growth and learning. I've been fortunate enough to carve out a unique professional niche to protect the environment that I call home. There are several people without whom this would not have been possible.

Firstly, I'd like to thank NatureLink Perth for providing the platform and community of practice that enabled this research. Thank you to my supervisors, Drs Jane Chambers and Margaret Andrew, for offering your expertise and professional experience, your many hours of reviewing, your creative ideas in solving problems I wasn't sure how to approach, and challenging me to push my understandings. Jane's local ecology experience and Margaret's GIS wizardry were extremely valuable in guiding this thesis.

To my family, thank you for offering endless love, support and understanding to any challenges I faced. Mum and Gran, you are my role models and inspire me to be a better problem-solver, communicator, and person. Mitch and Bronte, thank you for supporting me through long days and nights of working, offering advice, and doing the little things every day that made my life easier. They didn't go unnoticed.

Thank you to my friends, who each helped in their own way. Demi, who offered great advice, especially on the best paths through formal processes and bureaucratic hoops. You saved me a lot of time, money, and headaches Demi – you're the best! Cody, who joined me for field surveys, shared good music and good company, and followed me on my complicated ecological reflections. I appreciate you Mr. Beans. An eclectic mix of other friends and colleagues – Sam, Auben, Benny, Spence, Harry, Em, Roux, Cev, Shav, Emma, and more – consistently checked in on my progress, offered advice or help, or just protected my overall sanity over the last two years. Much love to you all.

I also want to thank those from Murdoch University who supported me in this degree. Sean, who was of immense help guiding me through PRIMER and the associated analyses. Mate, I owe you a month of YouTube premium. Kimberly, for calmly turning every administrative problem into a solution. Finally, Christina O'Donnell, whom I have not met, but whose study developing Perth's naturelink network was foundational in shaping this work.

ABSTRACT

Urban biodiversity conservation depends on functional connectivity, yet most cities lack tools to translate broad connectivity principles into spatially explicit rehabilitation actions. This thesis develops a multi-scale, multi-species framework that identifies where rehabilitation will most effectively enhance connectivity in an extensively fragmented biodiversity hotspot – the Perth-Peel region of Australia's South West.

At the regional scale, rehabilitation priorities were ranked for 1268 ecological linkages using a multi-criteria decision analysis integrating conservation values, vulnerability, connectivity, and route attributes. High-priority linkages were identified across the region, with priorities driven more by the degree of landscape modification around each linkage than the conservation values and vulnerability of protected areas. The analysis discovered abundant rehabilitation opportunities for whole-network stability in rural landscapes, while highlighting the need to act on the limited urban opportunities, heavily driven by landscape constraints. However, linkage-specific planning remains essential.

At the local scale, on-ground rehabilitation priorities were identified using Circuitscape connectivity models that included wildlife guild-specific dispersal limitations. These methods were applied to three case studies containing high-priority linkages across different development contexts. For guilds with limited dispersal abilities (<500m), the outputs identified fine-scale movement routes and areas of functional disconnection, enabling targeted rehabilitation recommendations such as stepping stone creation, improvement of matrix permeability, or enhancing movement redundancy. The approach was less informative for long-range guilds.

Together, these two studies demonstrate the value of multi-scale connectivity planning. The work delivers a set of integrated decision support tools for practitioners to target where rehabilitation is of most benefit to connectivity, relevant to planning realities in Perth-Peel and wider Australia. Designed to be repeatable in other contexts, this work addresses a critical implementation gap and offers a pathway to translate connectivity theory into targeted, on-ground action.

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LIST OF ACRONYMS

General	
ANOVA	Analysis of variance
CALM Act	Conservation and Land Management Act 1984
CHM	Canopy height model
EP Act	Environmental Protection Act 1986
EPBC Act	Environmental Protection and Biodiversity Conservation Act 1999
GI	Green infrastructure
GIS	Geographic information systems
LCP	Least-cost path
MCDA	Multi-criteria decision analysis
NDVI	Normalised difference vegetation index
PA	Protected area
PCA	Principal component analysis
PERMANOVA	Permutational multivariate analysis of variance
SIMPER	Similarity percentages (analysis)
Tukey's HSD	Tukey's honest significant difference
WA	Western Australia
WoRMS	World Register of Marine Species
Organisations	
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DBCA	Department of Biodiversity, Conservation and Attractions
DCCEEW	Department of Climate Change, Energy, the Environment and Water
DPIRD	Department of Primary Industries and Regional Development
DPLH	Department of Planning, Lands and Heritage
ESA	European Space Agency
GANSW	Government Architect New South Wales
NRMMC	Natural Resource Management Ministerial Council
WALGA	Western Australian Local Government Association
WAPC	Western Australian Planning Commission

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CHAPTER 1:

GENERAL INTRODUCTION

1.1 Background

Urban environments are misunderstood. Their outward appearance paints a picture of ecological degradation – an environmentally simplified landscape with little value to biodiversity and conservation. To an extent, this is true. Urban development traditionally clears large amounts of land, exploits natural resources, severely fragments and alters habitats, and presents large challenges to restoration (Ingram, 2008; Newman and Jennings, 2008; Ikin *et al.*, 2015; LaPoint *et al.*, 2015). Yet paradoxically, these same landscapes can be among the most important places for conservation. Cities are often built on areas of high biodiversity, as these areas historically offered ample natural resources and ecosystem services crucial for the establishment of settlements (Newman and Jennings, 2008; Ives *et al.*, 2016). Today, urban biodiversity continues to supply ecosystem services to humans and forms the basis of most human-nature interactions, as the majority of people now live in cities (Luck *et al.*, 2011; Ahern, 2013; Beninde *et al.*, 2015). Urban areas also contain a disproportionate share of threatened species. In Australia, cities cover just 0.23% of the terrestrial area yet support around 30% of the nation's threatened species (Ives *et al.*, 2016). In some cases, cities support the last remaining populations of endemic species, making urban conservation essential to preventing extinctions (Soanes and Lentini, 2019).

Cities are inherently subject to extensive habitat clearing and fragmentation, and a once continuous natural environment often becomes reduced to remnant patches that are too small and isolated to provide sufficient resources to support stable populations of native flora and fauna (Ramalho *et al.*, 2018; Ottewell *et al.*, 2019). This is evident in the Perth-Peel region of Australia's South West. A city not only built in a globally-recognised biodiversity hotspot, but built in the most species-dense portion of this region (Gioia and Hopper, 2017), Perth faces ongoing challenges with clearing, fragmentation, and population instabilities in remnant habitats (How and Dell, 2000; Davis *et al.*, 2013; Ramalho *et al.*, 2018). The heavily modified urban context can make nature conservation and repair challenging, but not impossible (Ingram, 2008; Lynch, 2018). With strategic planning, ecological rehabilitation can be used to promote ecological connectivity and species movement between core habitats, allowing populations to access sufficient resources to survive (FitzGibbon *et al.*, 2007; Braaker *et al.*, 2014; Croeser *et al.*, 2024; Newell *et al.*, 2025). However, effectively applying these strategies first requires an understanding of what connectivity and rehabilitation entail in an urban setting. A summary of these concepts is provided below (Box 1) and are discussed in more detail in Chapter 2.

Box 1: Key concepts

Ecological connectivity refers to the degree to which the landscape facilitates movement among resource patches (Taylor et al. 1993). Two fundamental approaches exist to define and measure connectivity: structural connectivity, which refers to the physical arrangement of landscape elements that facilitate connectivity, while functional connectivity is species-focused and considers species behaviour in the landscape (Keeley et al. 2021). This thesis focuses primarily on functional connectivity, though it utilises 'potential' rather than 'actual' connectivity metrics, using models which make assumptions about species behaviour rather than measuring on-ground data (Calabrese and Fagan 2004).

Ecological rehabilitation is the process of repairing or enhancing ecological function (Choi 2007). In contrast to restoration, which aims to restore natural ecosystems based on a historical reference (Gann et al. 2019), rehabilitation prioritises improving ecological processes to a variety of extents. In the context of enhancing connectivity (i.e. an ecological process) in human-modified environments, rehabilitation provides flexibility in creating habitat value in diverse ways across a range of urban contexts, while avoiding limiting itself to the potentially unrealistic ambition of a historical reference (Hobbs et al. 2009; Klaus and Kiehl 2021).

1.2 Research context

The current state of conservation research recognises the critical role of functional connectivity in enhancing gene flow (Munshi-South, 2012; Ottewell *et al.*, 2019), maintaining and improving diversity (Shanahan *et al.*, 2011; Braaker *et al.*, 2014), enabling species to meet minimum habitat area requirements (FitzGibbon *et al.*, 2007; Peterson and Andrew, 2025), and supporting metapopulation stability in fragmented human landscapes (Taylor *et al.*, 1993; Hanski, 1999; Heard *et al.*, 2012; Fletcher *et al.*, 2016). Research also recognises the growing limitations and challenges associated with traditional conservation and restoration in enabling effective conservation outcomes in cities (Soanes and Lentini, 2019; Hobbs *et al.*, 2009). This realisation has spurred a shift in urban restoration practice, with practitioners shifting from a focus on historical fidelity to pragmatic rehabilitation strategies which prioritise ecological function in a range of urban contexts (Garrard *et al.*, 2018; Klaus and Kiehl, 2021; Soanes *et al.*, 2023).

Despite established research, an implementation gap remains in coordinating green infrastructure planning to support connectivity. In Perth-Peel, research has mapped regional ecological linkages (WALGA, 2004; O'Donnell, 2020), but we now need strategic guidance on where to focus limited resources and how to translate the regional network into local implementation, which remain challenges for effective connectivity planning (Norton *et al.*, 2016; Threlfall *et al.*, 2019). Alongside practical limitations, green infrastructure research for biodiversity is still in its infancy in Australia, with specific research gaps recognised for optimising its spatial configuration (Lin *et al.*, 2019; CSIRO, 2021). Research-based decision support can therefore be valuable for urban planning practice, which often lacks time, resources and science-backed methods for effective environmental decision-making (Ahern, 2013; Zuniga-Teran *et al.*, 2020; Patterson *et al.*, 2023). While emerging tools in Australia are beginning to address multi-species functional connectivity (e.g. Kirk *et al.*, 2023) and landscape-scale connectivity planning (e.g. Norman and Mackey, 2024), there is a distinct scarcity of integrated, multi-scale frameworks to bridge the gap between connectivity theory and planning practice.

1.3 Thesis outline

This thesis addresses this critical implementation gap. The aim was to develop a multi-scale, multi-species framework to spatially prioritise rehabilitation in the Perth-Peel region to optimise connectivity outcomes (Figure 1). The research builds on a regional ecological network developed by O'Donnell (2020), which identified ecological linkages between protected areas using least-cost path modelling. The approach in this thesis was twofold:

Firstly, multi-criteria decision analysis (MCDA) was used in conjunction with geographic information systems (GIS) to identify and rank the regional ecological linkages for rehabilitation across Perth-Peel, based on a range of environmental criteria. These criteria captured both the need and feasibility of rehabilitation to enhance connectivity, including the conservation values and vulnerability of the protected areas being linked, as well as the current connectivity and route attributes of the linkages themselves.

Secondly, a local-scale decision-support tool was developed to identify on-ground rehabilitation priorities for four wildlife guilds (tree birds, woodland birds, vertebrate terrestrial fauna, and vertebrate wetland fauna) with differing habitat requirements and movement capacities. The tool used outputs from circuit theory connectivity models to identify both spatial priorities and associated management recommendations. It was applied at three sites containing high-priority linkages from the regional MCDA, providing case studies across contrasting development contexts (existing urban, emerging urban, and rural) that reflected the diversity of the urban landscape. Analyses were supported by GIS and rapid field surveys, and the methods are repeatable in other urban areas.

The research contributes to connectivity planning in both research and practice. It acknowledges current research gaps and priorities in Australia (Lin *et al.*, 2019), demonstrating methods to spatially optimise green infrastructure planning. In practice, the research provides spatial decision support to assist planners, managers and conservationists, empowering them to make informed decisions relating to planning policy and strategically target local interventions to enhance regional connectivity.

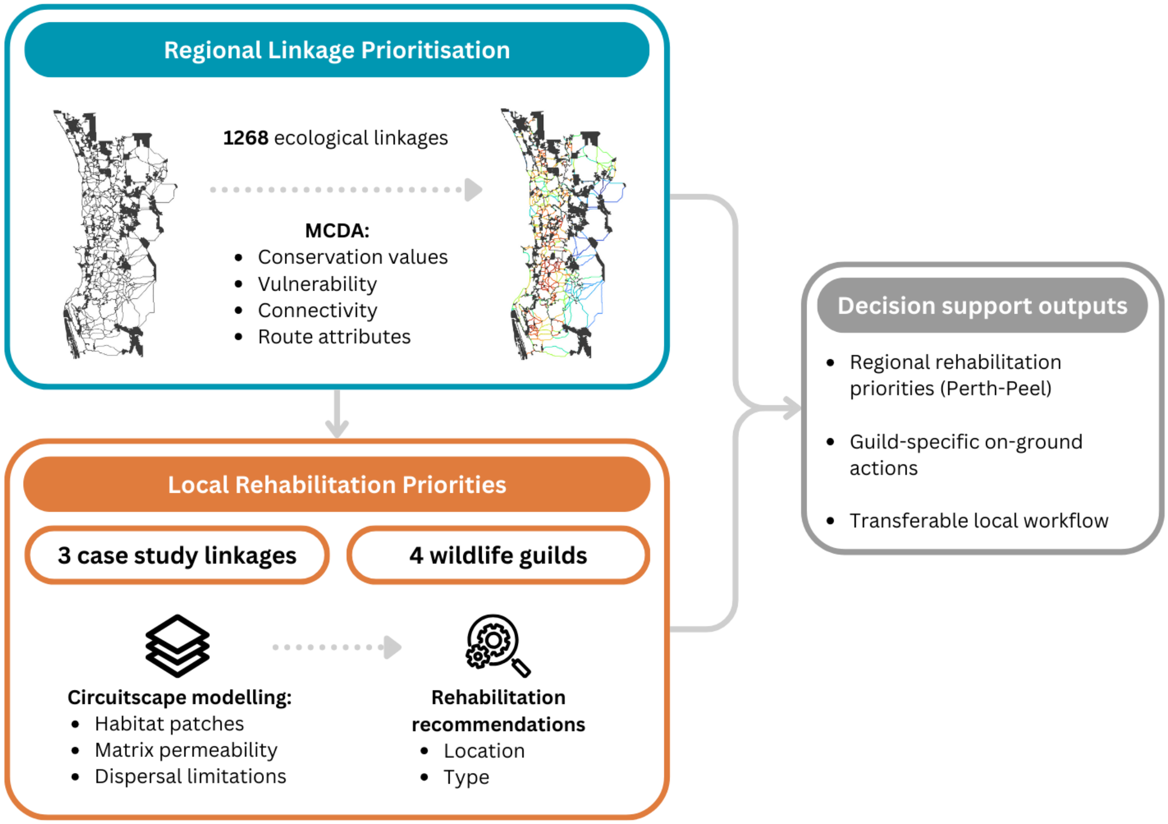


Figure 1. Workflow of the framework, linking regional-scale ecological linkage prioritisation with local Circuitscape modelling to produce a set of decision support outputs for connectivity planning.

1.3.1 Thesis structure

This thesis is presented in five chapters. Following this general introduction, Chapter 2 is a literature review on landscape ecology, rehabilitation, and green infrastructure planning in urban contexts, providing background information and justification for the following chapters. The core research is presented as two studies conducted at different planning scales but connected through their shared focus on the ecological linkage network. Chapter 3 is at the regional scale, prioritising 1268 ecological linkages across the Perth-Peel region to establish the broader network context for planning. Chapter 4 is at the local scale, applying a multi-species framework across three case studies to identify on-ground rehabilitation priorities from circuit theory functional connectivity modelling. Chapter 5 provides a general discussion, integrating local and regional scales into key findings and suggesting directions for further research.

CHAPTER 2:

LITERATURE REVIEW

This literature review examines the value of enhancing connectivity through ecological rehabilitation as an essential strategy for conservation within cities, with a particular focus on Perth, Western Australia. It first provides insights from landscape ecology to understand how urban biodiversity might best be sustained. The review then explores the inherent difficulties in urban ecological restoration, highlighting the need for a rehabilitation approach that makes use of a diversity of rehabilitation strategies to enhance habitat for wildlife connectivity within the urban environment. Finally, it synthesises these understandings to explore green infrastructure (GI) design and planning for biodiversity, emphasising the key considerations for spatial prioritisation of GI for this purpose. Through a critical analysis of existing literature, this review seeks to highlight how coordinated planning can incorporate landscape ecology and ecological rehabilitation to improve connectivity in cities, and to foster sustainable and resilient urban ecosystems.

2.1 Landscape ecology for urban biodiversity

2.1.1 Landscape ecology

To plan for biodiversity in the urban context, we need to understand it from an ecological perspective. While traditionally landscape ecology has not been commonly studied in urban areas (Young *et al.*, 2009), applying its theories to urban planning enables us to see and respond to spatial biodiversity patterns, enabling urban sustainability and conservation (Wu, 2008). Patterns in landscape ecology can be viewed through the patch-matrix model (Figure 2). This model, which sees the landscape as patches of habitat connected by corridors or dispersal through an uninhabitable matrix, is a foundational theory in landscape ecology (Lausch *et al.*, 2015) and enables us to describe various spatial patterns in biodiversity. While the binary between habitat and uninhabitable matrix that it presents has limitations (Norton *et al.*, 2016), the patch-matrix model is well-suited to urban environments like Perth which are dominated by the fragmentation of remaining habitats into small, disconnected patches (Davis and Glick, 1978; Stenhouse, 2004).

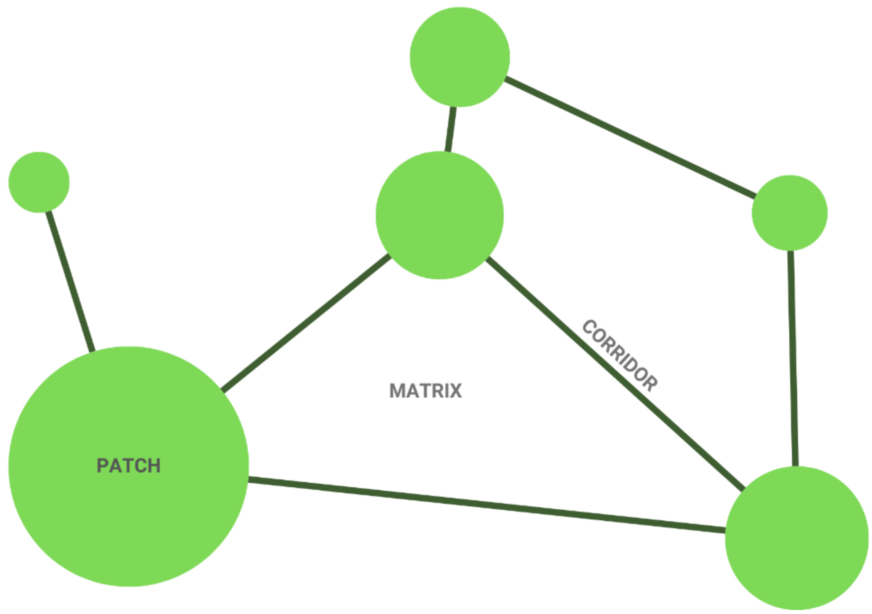


Figure 2. Conceptual diagram of the patch-matrix model; illustrating habitat patches of varying sizes (light green), connected by corridors (dark green) within a surrounding matrix (white background).

Furthermore, while in reality the matrix is variable and not uniformly hostile to species, the patch-matrix model is still able to account for this by differentiating land types within the matrix (Kumar and Cushman, 2022). For example, resistance-based connectivity analyses such as least cost path (LCP) or circuit theory model the flow of organisms across a 'resistance surface', representing the relative cost of movement across different land covers, to predict the most likely movement paths of wildlife between habitat patches (Cameron *et al.*, 2022).

Through clearing and land development for human use, urban environments are dominated by habitat loss and fragmentation. Landscapes that have fewer, smaller, and more isolated habitat patches support less biodiversity (Table 1), and unfortunately, in cities, where land is under pressure for multiple competing purposes, the scope for significantly increasing the area of habitat and habitat patches is limited (Klaus and Kiehl, 2021). A significant contribution to urban conservation, therefore, might be to reconnect habitats by promoting movement of biodiversity through the city, thereby increasing the total area of habitat that is accessible to a given organism.

To do this we need to understand and address the key factors influencing biodiversity in urban landscapes. Table 1 outlines these elements, as identified through landscape ecology, providing insight into the need to optimise habitat area, patch size, condition and connectivity, through sustaining habitat patches and reducing matrix hostility.

Table 1. Factors affecting urban biodiversity, as identified from a landscape ecology perspective. Factors are grouped by patch characteristics (the qualities of individual patches), and landscape structure (the structure of the entire landscape, include type and amount of habitat and their spatial arrangement). Each factor includes a definition where required and an explanation as to how it impacts biodiversity.

Factor	**Explanation**
Patch characteristics	
Patch size	• Patch area is consistently a top predictor of species diversity, with larger patches supporting higher species richness (Beninde *et al.*, 2015). • Patch size effects are supported by foundational theories of species area relationships and island biogeography (Arrhenius, 1921; MacArthur and Wilson, 1967). • This does not necessarily mean there will be more species per unit area, or that larger patches are more important. In fact, smaller patches often have disproportionate conservation value. This is because species accumulation flattens with increasing area, meaning they can support a higher average species richness per unit area, and they often represent the last remnants of specific communities in urban landscapes (Wintle *et al.*, 2019).
Patch shape	• Increasing patch shape complexity and patch perimeter:area ratio can decrease species richness due to edge effects (Norton *et al.*, 2016). See fragmentation below.
Landscape structure	
Total habitat area and habitat loss	• The total amount of habitat area within the landscape and species richness are positively correlated (Fahrig, 2013). • A global literature review found habitat loss has significant negative effects on biodiversity (Fahrig, 2003). • The habitat amount hypothesis (Fahrig, 2013) counters island biogeography, stating that species richness is not a function of patch size/isolation, but one combined metric of total habitat area within the local landscape.

Matrix quality	• Refers to how well the matrix (non-habitat) composition supports or hinders ecological processes; for example, species movement (Baum *et al.*, 2004). • Higher urban intensities in the surrounding matrix are generally associated with lower patch-level species richness, though patterns vary among taxonomic groups (Norton *et al.*, 2016). For example, urban centre edge-to-centre gradients describe how urban centres generally harbour less biodiversity than peri-urban counterparts. These gradients may not be linear and are often linked to development intensity, land-use, or heterogeneity (McDonnell and Hahs, 2008; Müller *et al.*, 2013). • Vegetation in privately-owned land can greatly influence local biodiversity, representing the largest source of urban greenspace and providing habitat value (Müller *et al.*, 2013).
Fragmentation	• Fragmentation, defined as the separation of habitat as opposed to loss of area, is globally understood to negatively affect biodiversity (Haddad *et al.*, 2015). However, responses of individual species vary greatly depending on a complex interaction of factors including life-history, disturbance tolerance, and a range of other factors that are not yet well understood (Fahrig, 2003; Ibáñez *et al.*, 2014). • Cities can create extreme cases of fragmentation, as urban infrastructure provides a more hostile matrix than other land uses such as agriculture. This can result in more pronounced negative effects. For example, research in Perth indicates significant and rapid loss of habitat quality in smaller remnants of banksia woodland plant communities, and longer-term functional extinction debt in larger remnants (Ramalho *et al.*, 2014, 2018). • Fragmentation can also increase perimeter:area ratio of patches (Norton *et al.*, 2016).
Connectivity	• Defined as the degree to which the landscape facilitates movement among resource patches (Taylor *et al.*, 1993). • Connectivity between habitat patches can enhance gene flow (Munshi-South, 2012), increase diversity (Shanahan *et al.*, 2011; Braaker *et al.*, 2014), and enable populations to access enough habitat to survive (FitzGibbon *et al.*, 2007; Peterson and Andrew, 2025). These are all crucial for metapopulation stability in fragmented environments (Taylor *et al.*, 1993; Hanski, 1999). • Literature reviews show uncertainty related to the effects of connectivity on urban biodiversity, highlighting knowledge gaps and species-specific responses (Bailey, 2007; LaPoint *et al.*, 2015). However, Fletcher et al. (2016) found despite complications in measurement, effects on biodiversity are largely quantifiable and positive.

2.1.2 Ecological connectivity

A global meta-analysis on habitat fragmentation by Haddad *et al.* (2015) points to connectivity as the solution to landscape fragmentation. Connectivity can allow species to persist in habitat patches that are too small to maintain stable populations on their own. It is thus essential for metapopulation resilience and species conservation, and to maintain ecosystem services (Taylor *et al.*, 1993).

Connectivity is a commonly explored concept in urban conservation research, much of which highlights its implications for planning (Crooks and Sanjayan, 2006; LaPoint *et al.*, 2015). Australian studies explore its impact on species ecology including population dynamics and persistence (FitzGibbon *et al.*, 2007; Ottewell *et al.*, 2019; Rycken *et al.*, 2022; Ferronato *et al.*, 2024; Peterson and Andrew, 2025). Others have demonstrated how connectivity can be practically applied in conservation planning, to prioritise remnant habitats or inform greening interventions that support wildlife movement (Poodat *et al.*, 2015; Kirk *et al.*, 2018, 2023; Norman and Mackey, 2024). In Perth, regional ecological linkages have been developed as foundational work for understanding the broader network (WALGA, 2004; O'Donnell, 2020). However, these efforts remain largely regional in scope, and there is still a need for approaches that support strategic prioritisation and guide local implementation. This represents a significant opportunity to refine the application of connectivity for urban conservation planning.

When it comes to measuring connectivity, there is a dizzying array of metrics to choose from (Keeley *et al.*, 2021). However, there are two fundamentally different ways that connectivity can be measured: as structural or functional connectivity. Structural connectivity describes the physical arrangement of habitat within the landscape based on summaries of the distances between patches. Functional connectivity relates this to the movement abilities of organisms and describes the how well the landscape facilitates the dispersal of different species (Keeley *et al.*, 2021). Lynch (2018) suggests purely structural metrics do not fully describe connectivity, as species' characteristics are just as important as landscape structure. They highlight that structurally isolated patches may be functionally disconnected for some species, but connected for other species that can traverse the matrix between them (Lynch, 2018). This has implications for conservation. If we wish to more tangibly link our conservation efforts with the characteristics of species we are trying to benefit, we require a shift to species-specific connectivity.

2.1.3 Species-specific connectivity

Species-specific connectivity offers a more nuanced view of how connected a landscape is, and how it is affected by traits specific to a particular species or group of species (Figure 3). Three interrelated traits contribute to connectivity:

1. *Habitat requirements* affect connectivity in two ways. They first define both the location and extent of suitable habitat, which for some species might be common, while for others is quite distinct. For example, Perth's black cockatoos (e.g. *Calyptorhynchus banksii naso*, *C. latirostris*) require mature native trees with hollows for nesting, and can find roosting and foraging habitat in a variety of trees all around Perth (Saunders *et al.*, 2014). However, Perth's southwestern snake-necked turtle (*Chelodina oblonga*) is dependent on wetlands for both food and shelter (Santoro *et al.*, 2024). While this affects the connectivity of a given landscape depending on the species, it does not refer specifically to functional connectivity, which describes movement between habitat patches. Habitat requirements do affect functional connectivity by defining matrix permeability, or how traversable the matrix is for a given species. A landscape is much more traversable if it provides key habitat elements such as food or shelter to support safe movement from one habitat patch to another (Baum *et al.*, 2004; Lynch, 2018).

2. *Movement ability* affects connectivity as it defines how far and by what means a given species travels through the matrix. The threatened Carnaby's black cockatoo (*C. latirostris*) can fly over barriers and can travel many kilometres between resources (EPA, 2019). However, small ground-dwelling animals such as quenda (*Isoodon fusciventer*) or the snake-necked turtle (*Chelodina oblonga*) can only travel in the realm of a few hundred metres, and are impacted by small barriers such as traffic-heavy roads (FitzGibbon *et al.*, 2007; Bartholomaeus, 2016; Santoro *et al.*, 2020).

3. *Sensitivity to urbanisation* influences both the suitability of habitat patches and the willingness of the species to traverse the matrix, both of which can affect connectivity. McKinney (2002) categorises species as urban exploiters, adapters, and avoiders. Urban exploiters utilise and even thrive in the urban environment, making habitat plentiful and

the matrix permeable. Urban adapters may utilise and traverse through anthropogenic landscapes under certain conditions, while urban avoiders require protection from them in large habitat patches. Given urban exploiters are likely to persist in even harsh urban conditions, and urban avoiders are likely to be restricted to remnant habitats (LaPoint *et al.*, 2015; Lynch, 2018), connectivity planning in cities holds its greatest conservation potential in supporting urban adapters, who are most likely to respond to designed habitat interventions.

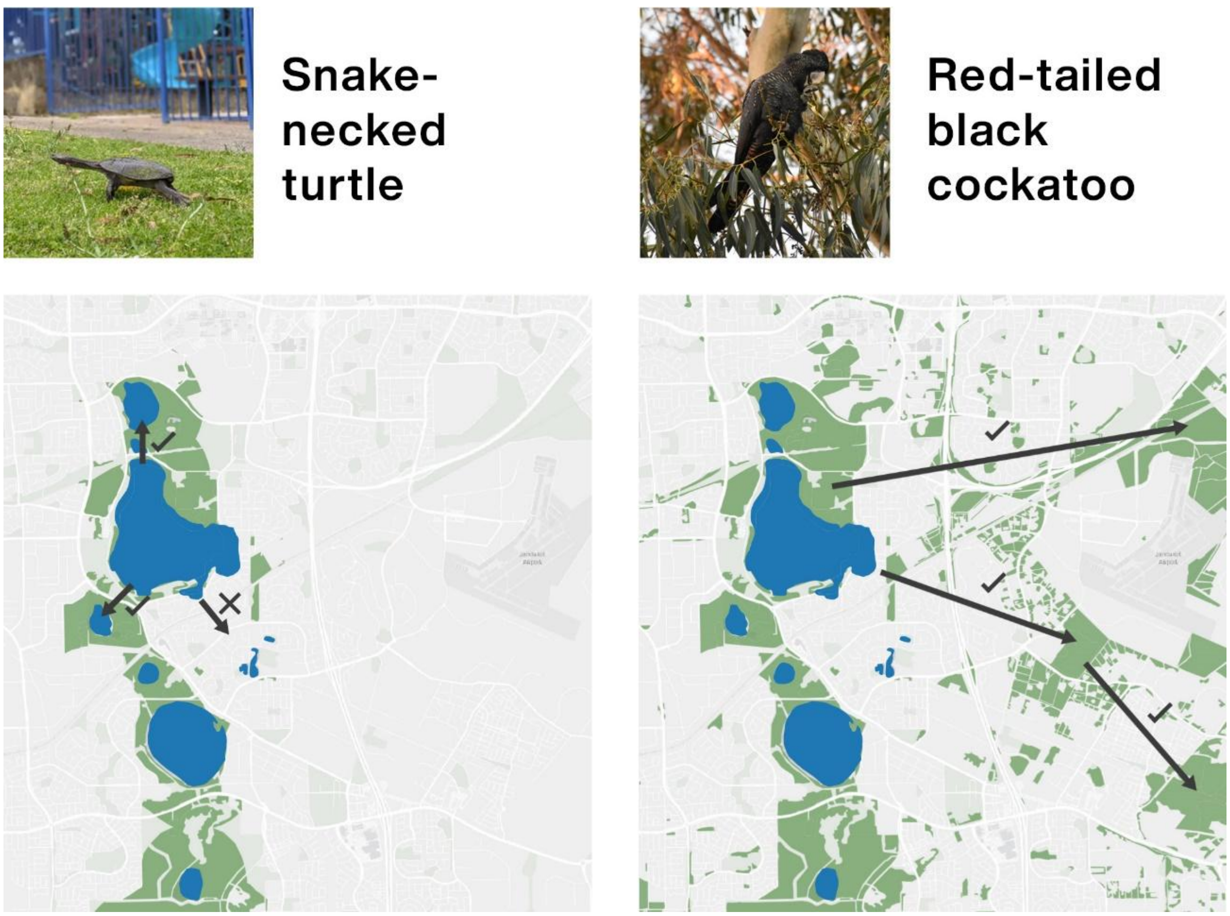


*Figure 3. Differences in connectivity between the southwestern snake-necked turtle (*Chelodina oblonga*) and red-tailed black cockatoo (*Calyptorhynchus banksii naso*). Potential habitat for each is shown in green (vegetation) and blue (wetlands). The snake-necked turtle has an approximate movement capacity of 500m (Bartholomaeus, 2016) as represented by the arrow length, while the black cockatoo can travel up to approximately 6km (EPA, 2019). Note the snake-necked turtle is unable to reach a wetland to the right of its habitat. Snake-necked turtle image: SOSNT 2024. Basemap sources: Esri, Geoscience Australia, NASA, NGA, USGS, TomTom, Garmin, METI.*

When planning for connectivity, a species or species group should be chosen to explicitly define what is and is not habitat to inform clearer decision making. Kirk *et al.* (2023) list three factors which should be considered when choosing focal species for connectivity planning:

1. Species distributions and local populations: species should be present.
2. Response of the focal species: species should be able to persist in urban environments under the right conditions. In short, the species should be urban adapters.
3. Planned actions: species should be impacted by the proposed planning interventions.

It is also important to select species whose ecology is well-studied, so that their habitat and movement requirements are known. Often species that are charismatic or are easily observed and identified are commonly studied (Garden *et al.*, 2006).

Selecting a species group or 'guild' with similar habitat needs and dispersal abilities, such as woodland birds or small mammals, can also be beneficial. Beyond compensating for potential lack of single-species data (Kirk *et al.*, 2023), a multi-species approach strikes a balance between being too general and too specific. Planning with structural connectivity is too broad to relate to the actual movement of any species (Lynch, 2018), while connectivity planning based on single-species does not necessarily confer benefits to other species, and may even result in dispersal barriers (Krosby *et al.*, 2015; Liang *et al.*, 2023). However, choosing an 'umbrella species' – one whose conservation protects a range of co-occurring species – can be an effective alternative (Breckheimer *et al.*, 2014).

2.1.4 Improving connectivity

Connectivity can be improved in cities through enhancing or creating habitat in four areas: habitat patches, corridors, stepping stones, and the matrix (Figure 4). Habitat patches are the backbone to connectivity, providing the sources and destinations of movement, refuges with essential resources and genetic sources of populations potentially traversing the urban environment (Hercock, 1997; Soga *et al.*, 2014). Enhancing these spaces can improve genetic connectivity and available habitat for shorter-ranging species (Ottewell *et al.*, 2019).

Corridors (linear) and stepping stones (small patches) are natural or semi-natural areas that provide some habitat value and facilitate movement but not necessarily long-term habitation. Unsurprisingly, ecological corridors are generally much more effective at increasing connectivity and species richness than stepping stones as they provide uninterrupted movement paths (Fahrig, 2003; Beninde *et al.*, 2015). However, continuous natural vegetation between patches is not always practical or achievable, especially in established urban environments. Stepping stones can be effective means of increasing connectivity in areas where corridors are not possible, but must be functionally connected to support animal dispersal (Baum *et al.*, 2004; Lynch, 2018). Additionally, the quality of the surrounding matrix influences matrix permeability, which in turn can improve the effectiveness of corridors and stepping stones (Baum *et al.*, 2004). Enhancing or creating habitat elements within the matrix, therefore, can also improve connectivity.

Figure 4. Aerial image showing four urban areas for which enhancing habitat can improve ecological connectivity, illustrating a habitat patch (outlined in light green), stepping stones (outlined in light yellow), and a corridor (outlined in light blue), within the surrounding matrix (the background landscape). Imagery © 2025 Google, Airbus, CNES, Maxar Technologies, Vexcel Imaging US.

2.2 Beyond traditional ecological restoration in cities

2.2.1 Challenges to urban restoration

Cities are unique and complex places to restore habitat for connectivity. Numerous and significant ecological and socio-economic challenges restrict restoration efforts (Figure 5). Traditionally, restoration has focused on recreating place-based historical species assemblages; to "mitigate abiotic change and reverse biotic change to push the system back towards historical, and more highly valued, composition and function" (Hobbs *et al.*, 2009). In many urban areas, however, restoration is often not undertaken to any degree as it is simply irreconcilable with the challenges involved (Ingram, 2008).

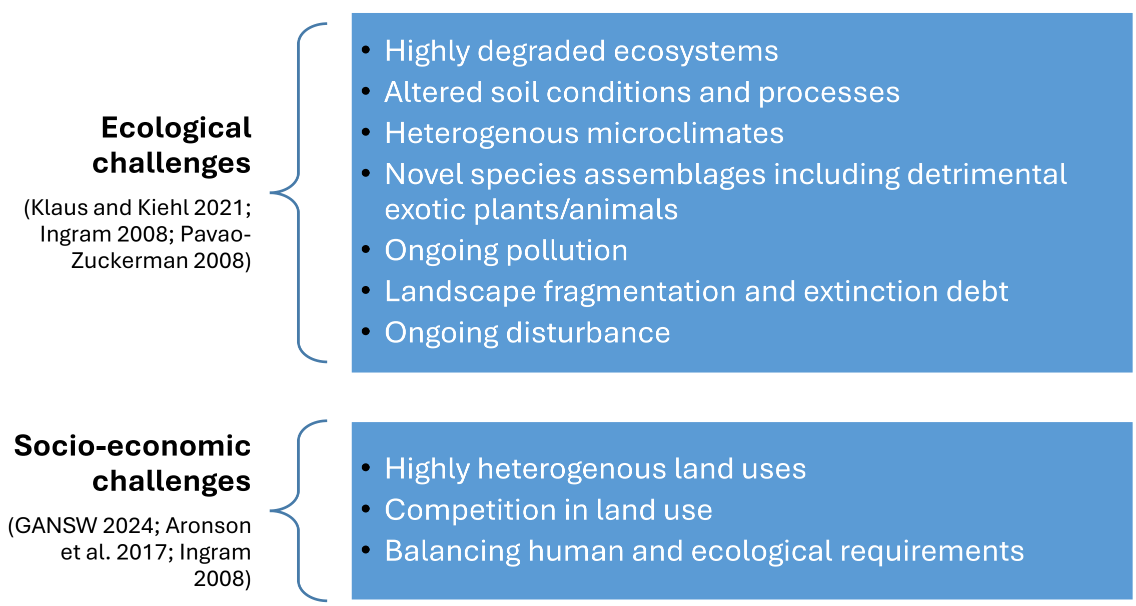


Figure 5. Urban challenges for restoration.

Cities have created and largely consist of what is termed 'novel ecosystems'. First formally articulated by Hobbs *et al.* (2006), novel ecosystems are systems which have undergone profound biophysical changes as a result of anthropogenic activity, and are now in a vastly different stable state from their historical counterparts. Such systems are often unable to be completely restored to their historical state as they have undergone irreversible ecological and biophysical change (Pavao-Zuckerman, 2008; Hobbs *et al.*, 2009; Perring *et al.*, 2013). Socio-economic factors often also impede a return to the original land use (Aronson *et al.*, 2017).

There is considerable controversy around the novel ecosystems concept in restoration, including concerns regarding what is considered irreversible and novel, and the potential bar-lowering implications for policy and management (Standish, Thompson, *et al.*, 2013; Murcia *et al.*, 2014; Miller and Bestelmeyer, 2016). There are some cases, therefore, in which connectivity planning must thoroughly and ethically discern whether a restoration approach is possible to avoid this downfall. However, in most instances, improving connectivity cannot rely on traditional restoration alone and we must move beyond traditional black-and-white thinking; utilising "novel concepts for multifunctional and biodiverse urban greenspaces" (Klaus and Kiehl, 2021).

2.2.2 Options for enhancing biodiversity

Options for creating habitat in cities are diverse and distinct, all offering unique opportunities for supporting urban biodiversity. In the urban restoration ecology literature, two fundamental approaches emerge (Table 2). The first is of course traditional restoration, which has its own suite of strategies that can be used in urban vegetation remnants. However, a second approach emerges with a key difference in its overall goal. Ecological rehabilitation emphasises ecological function rather than historical fidelity, meaning instead of trying to bring back the original ecosystem structure, function and composition, it aims to enhance the quality and functionality of the ecosystem (Choi, 2007). To improve connectivity, this may mean creating or enhancing specific habitat elements such as shelter or food, to fulfil species' needs and enable wildlife movement through the matrix.

A rehabilitation approach provides essential flexibility in enhancing biodiversity in urban contexts (Klaus and Kiehl, 2021). While necessarily 'lowering the bar' from an ecological perspective, rehabilitation unlocks the ability to integrate ecology with human-dominated urban environments – a practice called 'reconciliation ecology' (Rosenzweig, 2003). However, this also makes available a large variety of practical strategies in addition to those of traditional restoration. For example, habitat values may be improved by adding native vegetation and artificial habitat structures (e.g. nest boxes) to a park, vegetating a constructed wetland, or creating native verge gardens. Each strategy can benefit connectivity in unique ways (refer to Figure 4), and therefore planning should strive to utilise all strategies available to ensure habitat can be enhanced where it is most needed.

Table 2. Two fundamental approaches to ecological restoration in urban environments.

Approach	**Goal**	**Strategies in the urban context**
Traditional restoration	Restore natural ecosystems based on an historical reference (Choi, 2007)	• Restore core habitat (Standish, Hobbs, *et al.*, 2013; Ikin *et al.*, 2015). • Species reintroduction (Soanes *et al.*, 2020). • Spatial conservation and restoration planning in remnant habitat patches • People-led approaches – educate and engage community in restoration (Ikin *et al.*, 2015).
Rehabilitation	Repair or enhance ecological function (Choi, 2007)	• Utilise a more realistic urban reference system rather than an historical one; or target ecological structure/function (Perring *et al.*, 2013; Klaus and Kiehl, 2021). • Manage and support novel ecological values (Standish, Hobbs, *et al.*, 2013). E.g., exotic pine trees as foraging habitat for an endangered Perth cockatoo (Valentine and Stock, 2008). • Integrate ecology with urban planning and design through multifunctional GI (Ahern, 2007; Zhang *et al.*, 2019), nature-based solutions (Raymond *et al.*, 2017), or urban design for biodiversity (Beatley, 2011; Garrard *et al.*, 2018; Thomson *et al.*, 2022). • Create habitat analogues (Lundholm and Richardson, 2010). • People-led approaches – gardening for wildlife (Goddard *et al.*, 2010, 2013).

There are potentially many settings in which urban ecosystems can be rehabilitated to improve connectivity. The urban environment is highly heterogenous, and its different contexts each present distinct spatial and functional characteristics. As such, urban nature occurs in many forms, each with the unique potential to contribute to connectivity. A variety of urban typologies can support GI and contribute to ecological connectivity of urban landscapes (Table 3). For example, more linear typologies such as riparian corridors or undeveloped road reserves may support ecological corridors, while vegetation within campuses, plazas or private residences are often smaller, offering opportunity to act as stepping stones (GANSW, 2024).

Table 3. Urban typologies and the potential role that green infrastructure within each might play in improving connectivity. Adapted from GANSW (2024).

Urban typology	**Habitat patch**	**Corridor**	**Stepping stone**	**Matrix**
Campuses (schools, universities and hospitals)	•		•	•
Carparks			•	•
Cemeteries			•	•
Community gardens			•	•
Disused lots			•	•
Golf courses	•		•	•
Green walls and green roofs			•	•
Plazas and squares			•	•
Private gardens			•	•
Railway lines		•		•
Regional parks	•			
Reserves	•		•	
Riparian corridors	•	•		
Road verges		•		•
Urban parks		•	•	•
Wetlands	•		•	

We must take a holistic and opportunistic approach to enhance connectivity. Restricting ourselves to only one strategy or urban typology is to limit our ability for creative and adaptive decision-making required for conservation in a competitive environment (Aronson *et al.*, 2017; Soanes *et al.*, 2020). Rather than prescribing the specifics of implementation, the focus should be on identifying the ecological outcomes needed to improve connectivity. We need to understand 1) how to enhance habitat quality to support movement, and 2) where it is most needed. For this we need to explore GI design and planning.

2.3 Green infrastructure for wildlife – design and planning

2.3.1 Green infrastructure

In an urban context, GI is defined as all vegetation providing environmental, economic and social benefits (CSIRO, 2021). GI is an all-encompassing term for urban nature, from remnant bushlands to street trees, and in this thesis also includes 'blue' or 'blue-green' infrastructure including an aquatic component, such as drains or waterways. GI can also describe a network of natural and semi-natural areas, designed and managed to provide ecosystem services at multiple scales (Monteiro *et al.*, 2020).

While GI comes in many forms, it is an inherently human-centred concept and thus supports many goals. Along with biodiversity protection, the most studied aspects of GI over the last 30 years are human health, ecosystem service provision, and climate change adaptation (Ying *et al.*, 2022). GI is often multifunctional, designed to fulfil a combination of environmental, social and economic goals including urban heat mitigation, runoff regulation, recreation, or a combination of these (Ahern, 2007). Multifunctionality requirements mean design standardisation is not possible and the habitat created will look different with every implemented project. To provide for biodiversity, GI needs careful consideration of the type of habitat required and where it is needed most.

2.3.2 The building blocks of habitat

If designed well, GI has the potential to provide habitat and support wildlife movement. However, research on this is in early stages (Filazzola *et al.*, 2019) and benefits to biodiversity are likely to depend on how GI is implemented in practice (Salomaa *et al.*, 2017). To ensure success, GI design must account for the factors which influence habitat quality and thus the potential use of a site by wildlife.

Habitat quality is primarily shaped by site, vegetation, and disturbance-related factors (Table 4). It is crucial to take these factors into account when designing GI to ensure food and shelter provisions are present so it can realistically facilitate movement. Moreover, understanding a site's inherent opportunities and limitations, particularly concerning connectivity/isolation, legacy effects and current disturbances, can guide planning efforts to respond to these circumstances and maximise benefits or avoid pitfalls. For example, new GI offers the most value connecting and/or supporting remnant habitat, given the inherent biological richness of natural areas (Hercock, 1997; Filazzola *et al.*, 2019; Ottewell *et al.*, 2019). New habitats should thus be planned as supplemental to conservation of remnants and contribute to a connected ecological network, as GI created too isolated from these areas are less likely to support actual inhabitants or support connectivity. Additionally, areas with wetlands may provide greater potential for biodiversity based on the provision of water and diverse habitat types.

Table 4. Factors affecting habitat quality and biodiversity of green infrastructure.

Factor	Explanation
Site	
Proximity to remnant habitat	Proximity to remnant vegetation patches correlates to wildlife diversity, as the remnant habitat provides a source of biodiversity which will utilise surrounding resources (Braaker *et al.*, 2014; Soga *et al.*, 2014; Ottewell *et al.*, 2019).
Water sources and riparian vegetation	Vegetated water sources are biodiverse, provide many ecological niches and support aquatic and semi-aquatic wildlife (DBCA 2024; Beninde, Veith, and Hochkirch 2015). They are also often inherently connective elements in the landscape (Naiman *et al.*, 1993).
Vegetation	
Vegetation diversity	Plant diversity supports multiple ecological niches, and is a suitable indicator for wildlife diversity (Díaz and Cabido, 2001; Castagneyrol and Jactel, 2012).
Vegetation 'nativeness'	'Nativeness' of vegetation is an appropriate indicator for wildlife diversity in urban areas due to plant-animal evolutionary relationships (Berthon *et al.*, 2021).
Vegetation complexity	Vegetation density, cover and structure have significant positive effects on intra-urban biodiversity (Beninde *et al.*, 2015; Threlfall *et al.*, 2017).
Disturbance	
Historical disturbance	Legacy effects from past human disturbance can reduce a site's capacity to support biodiversity, as important biological and physical properties are irreversibly altered (Perring *et al.*, 2016, see Figure 4). As such, rehabilitation of natural or semi-natural systems is likely to support higher quality habitat and diversity per unit effort (Filazzola *et al.*, 2019).
Current disturbance	Areas subject to higher levels of human disturbance have reduced wildlife diversity, particularly specialist fauna species (Murphy and Romanuk, 2013; Bryant *et al.*, 2017).

As discussed previously, habitat requirements vary among species and taxonomic groups, and connectivity planning requires consideration of target species and their needs (Kirk *et al.*, 2023). To illustrate these nuances, a review on the ecology of Australian urban fauna by Garden *et al.* (2006) summarises drivers of urban diversity within five fauna groups – birds, mammals, herpetofauna, invertebrates, and aquatic species.

- **Birds:** Dependent on structurally and floristically diverse vegetation with a high proportion of locally native plant species; ideally being remnant habitat.
- **Mammals:** Influenced greatly by vegetation complexity. Mammal diversity is also more sensitive to connectivity between patches and human disturbances such as vehicle collisions and pet predation.
- **Herpetofauna (reptiles and amphibians):** Responses to urbanisation are more variable and less understood than other fauna groups, although native vegetation and water quality (for amphibians) are important factors for their diversity.
- **Invertebrates:** Can thrive in small and large patches alike, while vegetation diversity and habitat condition are important factors influencing their diversity.
- **Aquatics:** Factors impacting aquatic and semi-aquatic species diversity are not well understood, but it is dependent on water quality factors and is impacted heavily by drainage and runoff quality.

The benefits of knowing what constitutes habitat, both in general and for specific species groups, are twofold. First, it informs the key ecological outcomes needed from GI to best support wildlife, providing flexibility to achieve connectivity or biodiversity outcomes among other societal demands (Aronson *et al.*, 2017). Secondly, defining habitat provides insight as to where species are well-supported within the landscape and where they are not, which is essential for understanding connectivity. This leads to the next line of inquiry: where should habitat quality be enhanced?

2.3.3 Green infrastructure planning

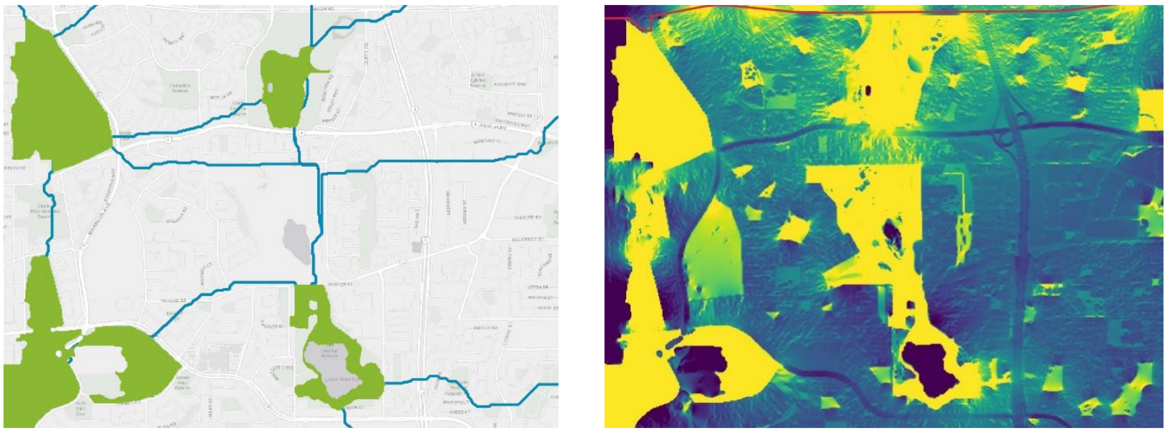
GI planning has been a successful way to incorporate landscape ecology principles into urban planning (Norton *et al.*, 2016). Particularly, it emphasises key principles of a multi-scale approach, pattern/process dynamics (i.e. biodiversity changes as a result of landscape structure), and the importance of connectivity (Ahern, 2007). While GI is often studied for its use in improving ecosystem services (e.g. Norton *et al.*, 2015; Longato *et al.*, 2023), there is a need to refine it from a biodiversity perspective. Urban planners often lack the resources and knowledge for effective conservation (O'Donnell *et al.*, 2017; Zuniga-Teran *et al.*, 2020), and research on GI for biodiversity in Australia is young (Lin *et al.*, 2019). Indeed, CSIRO (2021) cited increasing GI's capacity to support biodiversity and optimising GI spatial configuration as important research priorities. Given these challenges, GI planning with a focus on enhancing wildlife habitat and connectivity represents an important avenue for research and application.

There are some important factors to consider when prioritising GI for this purpose. Firstly, planning must consider to what extent enhanced connectivity is desirable in a given area. Some areas may support significant conservation value, such as species of conservation concern or rare habitats for which increased connectivity could improve resilience (Ahern, 2013). These considerations should be spatially quantified and used to weight prioritisation, particularly at a regional scale. Connectivity may also cause undesirable impacts where it helps to spread pests, weeds or diseases (Haddad *et al.*, 2014). However, research suggests evidence is scarce with relatively minor negative effects compared to connectivity benefits, while degraded habitat creates greater wildlife predation risks (How and Dell, 2000; Haddad *et al.*, 2014; Ottewell *et al.*, 2019).

Secondly, planning frameworks must address multiple scales to link regional-scale biodiversity processes and planning with local-scale processes and implementation. For example, introducing GI at a local scale is most effective if they are implemented in areas that are more important for regional connectivity, and thus contribute to the larger network (Qi *et al.*, 2021). Inversely, planning for connectivity at a purely regional scales may not provide the detail necessary to capture the heterogeneity of the urban environment and fine-scale movement paths which will inform implementation at local scales (Faeth *et al.*, 2011; Capotorti *et al.*, 2019; Donati *et al.*, 2022).

Thirdly, a multi-species approach is beneficial for holistic biodiversity planning, but entails additional considerations (Kirk *et al.*, 2023). It necessitates methods to map connectivity for separate species, which can be achieved by understanding the ecological requirements and movement abilities of particular wildlife groups (Kirk *et al.*, 2023). It also demands ways to combine results to infer multi-species priorities and effectively manage goals or trade-offs. A study by Molné *et al.* (2023), while only focusing on amphibian connectivity, normalised and combined the results of connectivity models for multiple frog species to infer multi-species GI priorities. This approach may also be applicable to multiple species groups.

Lastly, prioritising locations for new GI is more complex than prioritising existing assets. For existing GI, the relative value of each patch for connectivity can be readily quantified as patches are effectively fixed (e.g. Qi *et al.*, 2021). However, there may be numerous possibilities for new GI, and the connectivity benefits cannot be directly quantified, especially for each individual patch, without first being fixed in some form of rehabilitation scenario. While scenarios are valuable for evaluating improvements after sites have been selected (Kirk *et al.*, 2023; e.g. Croeser *et al.*, 2024), prioritisation must take a different approach. Resistance-based connectivity analyses such as least-cost path (LCP) or circuit theory have been utilised to identify leverage areas to improve connectivity (e.g. Balbi *et al.*, 2019; Liang *et al.*, 2023; Molné *et al.*, 2023). These approaches utilise a resistance surface to find the path(s) of least resistance between habitat patches, representing the most likely movement routes for organisms crossing the matrix (Zeller *et al.*, 2012). Setting cost values within a resistance surface, however, is a challenge that requires users to calibrate these values to achieve a satisfactory level of realism (Foltête and Vuidel, 2025; Merkens *et al.*, 2026). Where available empirical or modelled input data is limited, transparent reporting of parameter uncertainty is necessary, especially in applied conservation contexts (Zeller *et al.*, 2012). At the local scale, circuit theory is likely the more suitable resistance-based approach (Figure 6) as it effectively models more realistic diffuse movement patterns and provides more contextual information on movement quality, which is valuable for inferring local priorities for GI rehabilitation (McRae *et al.*, 2008; Pinto and Keitt, 2009; Cameron *et al.*, 2022; Kumar and Cushman, 2022).



Least cost path

- Identifies path of least resistance between habitat patches (Cameron *et al.*, 2022)
- Valuable for regional planning but can be somewhat one-dimensional at local scales, showing only one path between patches (Pinto and Keitt, 2009; Kumar and Cushman, 2022)
- Variation in movement along each path is not described

Circuit theory

- Provides a flow network that identifies areas of high and low current flow representing species movement (McRae *et al.*, 2008)
- Highlights relative importance of multiple contributing paths (McRae *et al.*, 2008; Cameron *et al.*, 2022)
- Describes movement quality throughout the landscape

Figure 6. Comparison of LCP and circuit theory approaches to identify local GI priorities in the urban landscape. The left image represents an example output of LCP analysis (O'Donnell, 2020), with habitat patches highlighted in green and LCPs in blue (basemap sources: Esri, Geoscience Australia, NASA, NGA, USGS, TomTom, Garmin, METI). The right image represents a circuit theory output of the same geographical area, with a gradient from dark blue to yellow representing low to high likelihood of movement (Image: Chambers et al.*, 2025).*

2.4 Conclusions

This review has explored the role of rehabilitation in enhancing connectivity for urban wildlife. Several critical findings emerge from the literature:

1. Urban landscapes are highly fragmented and lack capacity to significantly expand remnant habitat patches, so planning for multi-species connectivity between protected areas is essential for urban conservation.
2. Cities are novel ecosystems which make traditional restoration difficult. A rehabilitation approach which prioritises ecological function over historical fidelity is often more suitable in this context. Due to the diversity of rehabilitation approaches and urban typologies in cities, ecological outcomes of GI need to be the focus of rehabilitation, rather than taking a prescriptive approach towards implementation.
3. GI can enhance connectivity if its design and planning are informed by species habitat requirements and functional connectivity. However, spatial prioritisation for connectivity must consider various challenges including conservation values, multiple scales and species, and the methodology for assessing connectivity priorities at a local scale.

Despite increasing recognition of the importance of connectivity in urban planning, there is still limited integration of multi-scale, multi-species prioritisation approaches to GI in Australia. With a strong research demand on GI for biodiversity, further research is needed to refine spatial prioritisation methods and develop practical frameworks for planning connectivity-driven rehabilitation in cities that optimise biodiversity outcomes.

CHAPTER 3:

PRIORITISING REGIONAL LINKAGE REHABILITATION USING MULTI-CRITERIA DECISION ANALYSIS

3.1 Introduction

3.1.1 Context and aims

Optimising connectivity planning requires us to understand that ecological networks are inherently spatially heterogenous. Landscape structure varies across and within regions – including patch characteristics; total habitat area; matrix quality; and fragmentation – and so do patterns in biodiversity (Cadenasso *et al.*, 2007; Faeth *et al.*, 2011; Beninde *et al.*, 2015; Norton *et al.*, 2016). As a result, the need for connectivity and for improvements to connectivity is not uniform. Some areas may contain high biodiversity, or support species or communities of conservation concern. Some may be more heavily fragmented or contain vulnerable and isolated habitats. While the need for enhanced connectivity varies across ecological networks, so too does the feasibility. For instance, highly urbanised areas may limit the opportunities for rehabilitation or negate the capacity of species to move through it. Effective connectivity planning essentially depends on the underlying landscape to establish adequate ecological resources and support movement without prohibitive implementation challenges (Ingram, 2008; Aronson *et al.*, 2017).

An ecological network of 1268 linkages has been developed for the Perth-Peel region (O'Donnell, 2020) across a range of ecological and planning contexts. These linkages are potential movement pathways between protected areas intended to guide regional conservation planning, though they naturally vary in both their need and feasibility to support connectivity improvements. In this context, systematic methods are required to turn these varied linkages into actionable priorities where rehabilitation may produce optimal connectivity and biodiversity outcomes. Understanding how these priorities are distributed across the landscape, and which factors drive those patterns, is essential for interpreting where and why rehabilitation is needed. Spatial conservation prioritisation tools have traditionally been applied elsewhere to optimise ecological network planning based on biodiversity patterns (Watts *et al.*, 2009; Lanzas *et al.*, 2019; Jalkanen *et al.*, 2020), however, rehabilitation planning for network enhancement requires a method capable of integrating biodiversity with a range of connectivity need and feasibility factors. For this purpose, multi-criteria decision analysis (MCDA) emerges as an appropriate method. It has been a popular tool in nature conservation and spatial planning – with nearly 75% of nature-related MCDA studies focusing on management and restoration of natural areas, conservation prioritisation, and protected area planning (Esmail and Geneletti, 2018). Various studies have utilised MCDA specifically for spatial prioritisation of restoration, including for linear landscape features like streams or ecological corridors (e.g. Corsair *et al.*, 2009; Ferretti and Pomarico, 2013; Fernández and Morales, 2016; Cavalcante *et al.*, 2022). It provides a transparent and systematic framework for balancing diverse competing considerations and ranking the suitability of different options (Esmail and Geneletti, 2018), and offers useful information as to which considerations most influence suitability. This makes it well-suited to understanding rehabilitation priorities across the region's ecological linkages while balancing ecological value, need, and practical feasibility.

This chapter addresses this need for strategic decision making to enhance regional connectivity in the Perth-Peel region of Western Australia. Building on the established regional ecological network (O'Donnell, 2020), the analysis applies a broad spatial lens to identify where rehabilitation is best placed to support regional connectivity and why. Using MCDA, the study evaluates which regional linkages should be targeted for rehabilitation to maximise ecological outcomes. Specifically, this chapter aims to:

1. Determine which regional ecological linkages should be prioritised for rehabilitation to optimise wildlife connectivity benefits.
2. Understand the spatial distribution of priorities across the landscape, and to explore whether any linkage characteristics shape those patterns.

Outputs from this study establish the strategic foundation for local-scale connectivity modelling and rehabilitation planning, explored in Chapter 4.

3.1.2 Study area

The study takes place in the Perth-Peel region of Western Australia (WA) (DPLH and WAPC, 2015) over an area of 8058km^2 (Figure 7). The region spans 160km north to south, and extends approximately 50km inland from the coast, west to east. Nested within the Southwest Australian Floristic Region, the study area is in a globally recognised biodiversity hotspot (Myers *et al.*, 2000). This broader region contains an extraordinary diversity of biota, resulting from a complex combination of infertile soils, ancient landscapes, habitat complexity, and long-term climate stability (Hopper and Gioia, 2004). It contains around 8000 known plant species, over half of which are endemic to the region (Brundrett, 2021). Unfortunately, South West WA has undergone severe habitat loss, having lost nearly 90% of primary vegetation since European settlement, primarily for agriculture and urban development (Brundrett, 2021). This aligns with the internationally recognised criteria for global biodiversity hotspots (>70%) and this region was so declared in 2000 (Myers *et al.*, 2000). The Perth-Peel region itself has been less of a target for agricultural development than other portions of South West WA, retaining approximately 50% of remnant native vegetation (DPIRD, 2023), though an unknown portion of this is degraded. Although this extent of clearing is lower, the area is rapidly urbanising and habitat fragmentation remains a key element of landscape structure.

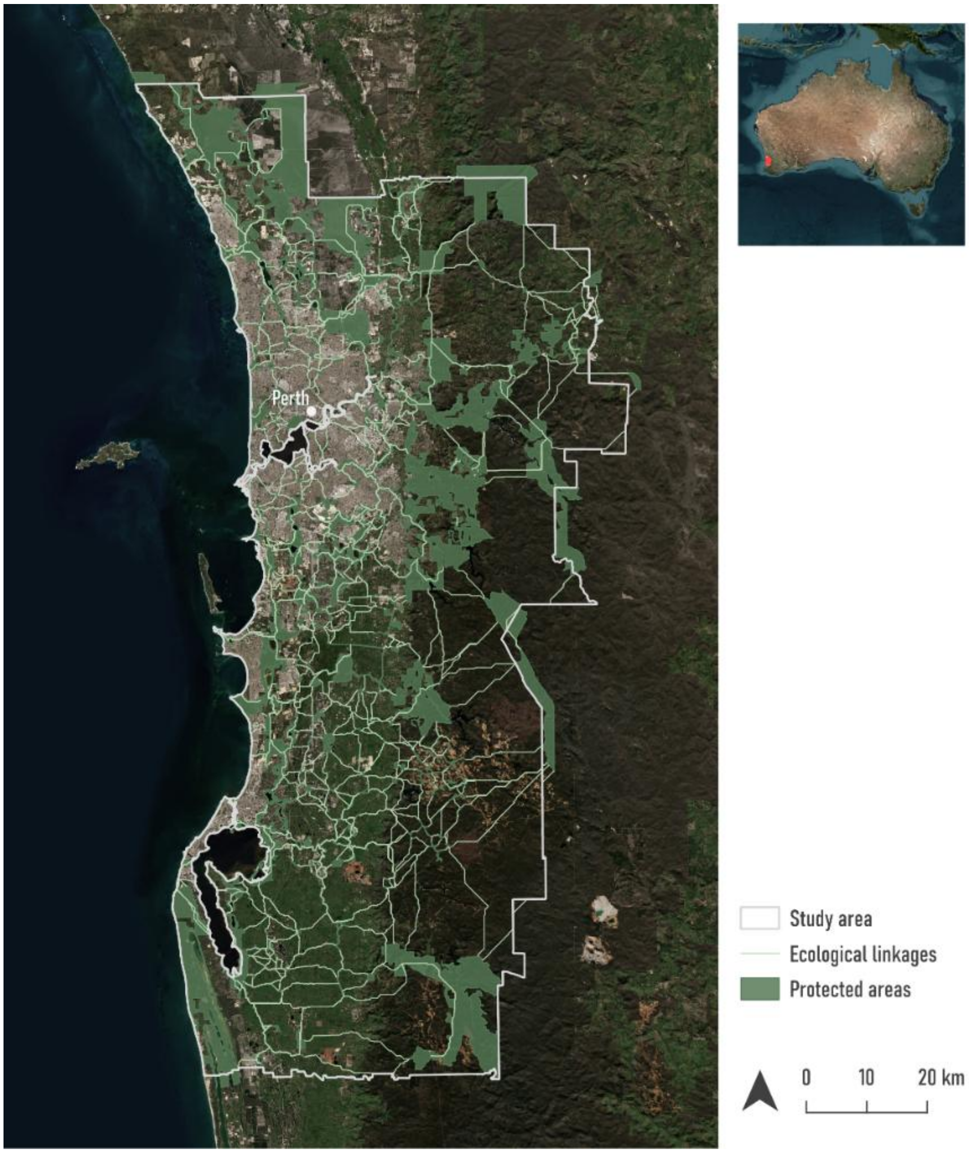


Figure 7. Map of the Perth-Peel region, showing the study area (white linework), its location within Australia (inset, top right), and the ecological network used for the analysis (O'Donnell, 2020). Base map from Esri (2025). Sources: Esri, CGIAR, TomTom, Garmin, FAO, METI/NASA, USGS, Earthstar Geographics.

The study area consists of two distinct bioregions. The Swan Coastal Plain, spanning the western half of the site, is a low-lying coastal plain mainly covered by banksia and eucalypt woodlands (DCCEEW, 2023). The plain supports diverse remnant bushlands and numerous wetland chains arising from an undulating linear dunal structure (Balla, 1994). However, it has been a primary area for urban and agricultural development (Figure 8), resulting in complex fragmentation effects on both plant (Ramalho *et al.*, 2014, 2018) and animal (How and Dell, 2000) communities. The Jarrah Forest bioregion spans the eastern half of the site, and is geologically distinct from the plain, sitting atop the Darling Scarp – a plateau of dominantly laterite gravels which support diverse eucalypt forests (DCCEEW, 2023). Apart from some agricultural clearing in the northeast of the site (Figure 8), the scarp remains largely intact of native vegetation, historically retained as water catchments for the city feeding several dams, although considerably modified in places by mining and scattered development.

Perth-Peel is a rapidly growing metropolitan region of over 2 million people. Its spatial footprint has increased 45% between 1990 and 2015, growing over 320km^2 (MacLachlan *et al.*, 2017). Largely uncontrolled urban sprawl and background infill development have accommodated growth (Hiller *et al.*, 2013; Bolleter, 2016) but continue to exacerbate habitat loss and fragmentation. Strategic planning frameworks such as Perth-Peel @3.5 million exist to guide future planning directions within the study area and balance development with conservation (DPLH and WAPC, 2015). However, existing land use planning in the study area is divided between the Perth Metropolitan Region and the Peel Region (Figure 8). Regional ecological linkages were formally developed for the Perth region in 2004 (WALGA, 2004), and have been recognised in local planning but not prioritised in practice, leaving a significant gap between strategic planning intentions and on-ground implementation. More recently, the linkages created by O'Donnell (2020) have been incorporated into several local government biodiversity plans. However, this more recent interest in connectivity planning is in its infancy, with limited resources and methods to prioritise planning efforts.

This combination of rapid expansion and limited operationalisation of ecological linkages creates a challenging landscape for managing regional ecological connectivity. However, prioritisation offers an opportunity to focus resources to strengthen the ecological network as the region continues to grow.

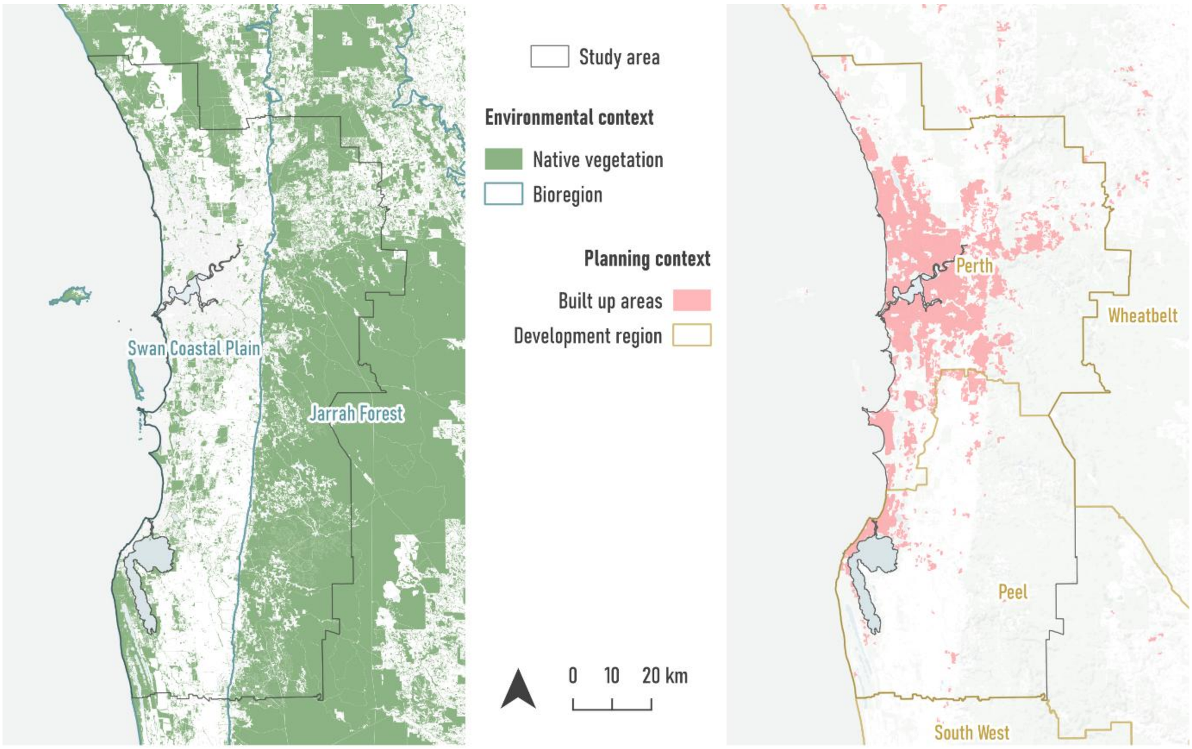


Figure 8. Planning and environmental context of the study area. Environmental context (left) shows remnant native vegetation in green, and bioregions outlined and labelled in blue; planning context (right) shows built up areas in pink, and development regions outlined and labelled in orange. The study area is outlined in black. Data: (DCCEEW, 2023; DPIRD, 2023, 2024; Geoscience Australia, 2023).

3.1.3 Ecological linkage network

This study builds directly on a regional ecological network created in 2020 (O'Donnell, 2020). The network consists of 420 protected areas (PAs), connected by 1268 ecological linkages (Figure 7). PAs were compiled from a range of datasets representing land with a level of environmental protection recognised at international, federal or state levels (Table 5). Most of the PAs are managed by the Department of Biodiversity, Conservation and Attractions, the primary state government conservation agency in Western Australia. However, there are a range of categories that differ in their management and strength of conservation focus, including Bush Forever sites, which are areas of regionally significant bushland and wetlands under varied jurisdictions that were strategically identified in planning policy of the early 2000's (Government of Western Australia, 2000). As a result, PA categories were classed as either 'formal' or 'semi-formal', based on the strength and clarity of the protection they receive for conservation from existing legislation (O'Donnell, 2020). While 'formal' PAs are those subject to clear legislation or managed by government authorities, the protection of 'semi-formal' PAs was obscured by variable land ownership, or uncertain proportions of area managed for recreational purposes (Government of Western Australia, 2000; Dooley *et al.*, 2011). Importantly, the final PA patches do not each represent a unique protected area (although they are henceforth referred to as such for convenience). Patches were post-processed by 1) clipping PAs to existing remnant vegetation cover, 2) simplifying geometries to remove holes and small gaps between patches of less than 50m, and 3) removing patches under 10ha (O'Donnell, 2020).

The ecological linkages represent least-cost paths between PAs. The linkage network connects each PA to up to its four closest neighbours, based on cost-weighted distances calculated using an underlying resistance (cost) layer. This resistance layer, developed at 10m resolution, represents how much the landscape enables or inhibits species movement, and integrated the unique impacts of land cover, land use, and barriers on species movement (O'Donnell, 2020). Each linkage therefore represents not intact ecological corridors, but potential routes of least resistance between PAs.

This regional network's ecological linkages and PAs were used as the basic analysis units for the MCDA, with the ecological linkages being the final unit of prioritisation. Other elements of this network were also utilised to characterise rehabilitation priorities, including the underlying resistance layer (O'Donnell, 2020).

Table 5. Datasets used for the protected areas within the Perth-Peel ecological network. Adapted from O'Donnell (2020).

Protected areas	Description	Protected area classification	Data source
IUCN Category 1–4 Lands	"A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (IUCN 2008)	Formal	(DBCA, 2025b)
National Parks	Protected by federal governments for the preservation of wildlife or human enjoyment	Formal	(DBCA, 2025b)
Nature Reserve	Land to preserve flora, fauna, and physical features	Formal	(DBCA, 2025b)
Conservation Parks	Land held by the Crown for conservation purposes	Formal	(DBCA, 2025b)
CALM Act Section 5(1)(g) reserves	Land vested in the Conservation and Parks Commission that is not a National Park, Nature reserve, or Conservation Park	Formal	(DBCA, 2025b)
Ramsar sites	Wetlands of international importance protected under the EPBC Act	Formal	(DBCA, 2017b)
Conservation Category Wetlands	Wetlands with a high level of ecological attributes and functions and are of highest conservation priority	Formal	(DBCA, 2024)
Botanical Parks Authority	For conservation of biodiversity and Botanic gardens	Formal	(DBCA, 2025b)
Regional Parks	Large areas recognised for conservation and recreation with multiple stakeholders	Semi-formal	(DBCA, 2023)
Bush Forever	Natural bushland identified as locally significant and requiring protection	Semi-formal	(DPLH, 2019)
Class A Reserve 1	Class A reserves protecting areas of high conservation or community value	Semi-formal	(DBCA, 2025b)
Class A Reserve 2	Class A reserves protecting areas of high conservation or community value	Semi-formal	(DPLH, 2023)

3.2 Methods

Multi-criteria decision analysis (MCDA) was used to prioritise rehabilitation of 1268 ecological linkages across the study area, based on a range of environmental and spatial criteria. The analysis included characteristics of both the linkages and the protected areas (PAs) they connect. The PAs are the primary conservation assets – the core habitats for which connectivity can enhance regional resilience and metapopulation stability (How and Dell, 1994; Soga *et al.*, 2014; Ottewell *et al.*, 2019). Conversely, the linkages represent the least-cost paths between these PAs – the potential movement pathways across the matrix. Including both units ensures that rehabilitation priorities reflect both the conservation priorities for the PAs, and the structural and functional relationships affecting connectivity between them.

Building the MCDA model occurred in two stages: defining and collating information for criteria to be used in the MCDA (Table 6) and then performing the MCDA. The latter involved determining criteria weights and calculating overall linkage priorities. The final analysis unit was therefore the linkage, with each linkage evaluated and ranked based on attributes of the linkage itself and of its two respective PAs.

3.2.1 Criteria preparation

Thirteen criteria were specified to characterise PAs and linkages in the study (Table 6). These criteria were organised into four categories representing different ecological and spatial dimensions of the prioritisation. *Conservation values* and *vulnerability* were used to prioritise PAs based on their ecological importance and on their size and degree of isolation within the regional network. For the linkages, categories included their current level of *connectivity*, which balanced the need for enhanced connectivity with the feasibility of facilitating adequate species movement; and *route attributes*, which identified opportunities and threats to rehabilitation along the linkage. The analysis deliberately excluded socio-economic influences to reduce complexity and establish a strong ecological basis for priorities in the region.

Six criteria characterised PA *conservation values*, including 1) the rarity of vegetation complexes present, 2) threatened ecological community extent and diversity, the richness of threatened 3) flora and 4) fauna, 5) wetland conservation values, and 6) the area of 'formal' protection within each PA (see Table 6 for detailed explanations). The criteria representing PA *vulnerability* included patch size and isolation, indicated by the area of remnant vegetation within the PA and an estimate of the connectivity of the PA based on its position in the regional ecological network (Table 6). Linkage *connectivity* criteria included length, average permeability (resistance to species movement), and the maximum distance between patches of remnant vegetation along the linkage as an indicator of habitat separation (Table 6). Finally, criteria describing the linkage *route attributes* included the number of barriers crossed, and an index of rehabilitation suitability constructed from the proportion, by area, of vegetation types within 100m of the linkage. Vegetation types included remnant vegetation, annual or non-remnant perennial vegetation, and bare ground or built-up surfaces. While all vegetation types were given different suitability scores, annual and perennial vegetation types were generally considered more suitable because they represent altered landscapes with uncertain species provenance, suggesting a potential need for rehabilitation. Remnant vegetation and bare surfaces were lower priorities, as remnant vegetation is likely to already provide habitat value, while bare areas such as paving, asphalt or buildings present substantial rehabilitation challenges.

As shown in Table 6 below, each criterion was represented by a measurable variable, which was calculated for either PAs or linkages using QGIS 3.38.2 (QGIS Development Team, 2025) and RStudio 4.4.1 (R Core Team, 2025). Detailed descriptions of how criteria were calculated are provided in Appendix A.

Table 6. Criteria chosen for the regional MCDA. Criteria fall within four categories that characterise the protected areas and ecological linkages. Each criterion is represented by a measurable variable, supported by a rationale, and informed by spatial or environmental datasets. Methods used to calculate each criteria are described in detail in Appendix A.

Category	Criterion	Variable	Datasets	Rationale
Protected areas				
Conservation values Identifies patches with high conservation significance and long-term biodiversity value	Vegetation complex rarity	Percentage remaining of vegetation complexes within the PA compared to pre-European settlement (area weighted mean of complexes present)	• South West vegetation complex statistics 2018 (DBCA, 2019) • Vegetation complexes – Swan Coastal Plain (DBCA-046) (DBCA, 2018b) • Vegetation complexes – South West forest region of Western Australia (DBCA-047) (DBCA, 2018a)	Rarer vegetation complexes are likely to represent higher conservation value as they are less widespread and more susceptible to further loss (Gaston, 1994).
	Threatened and priority ecological community (TEC/PEC) extent and diversity	Custom index based on the proportion of area covered (extent) and the number (diversity) of unique TEC/PEC types present within the PA (Appendix A)	• Threatened species and communities database (DBCA, 2025c)	The number and extent of TECs and PECs directly relates to conservation value under the *EP Act 1986 (WA)* and the *EPBC Act 1999 (Cth)*. A multiplier was included so that PAs that support multiple unique TECs/PECs are valued higher.
	Threatened and priority fauna richness Threatened and priority flora richness	Residuals of a linear regression of species richness by survey effort, used to correct for sampling bias. Negative values represent less species than expected given the sampling effort, with positive values representing more than expected.	• Threatened species and communities database (DBCA, 2025c) • Dandjoo database (DBCA, 2025a) • World Register of Marine Species (WoRMS) database (Ahyong *et al.*, 2025)	Threatened and priority species directly relate to conservation value under the *EP Act 1986 (WA)* and the *EPBC Act 1999 (Cth)*. Sampling bias correction is crucial for presence-only species data in which survey effort varies geographically (Bystriakova *et al.*, 2012).
	Wetland values	Highest wetland management category present within the PA, reflecting relative wetland conservation values assigned under the state wetland assessment framework (DBCA, 2025d)	• Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024)	Perth's wetlands provide critical habitat for a variety of birds, aquatic and terrestrial fauna, including many migratory and threatened species (DBCA, 2025d).
	Formal protection	Proportion of PA that is formally protected (see Section 3.1.3)	• NatureLink Perth PA data (O'Donnell, 2020)	Formally PAs are likely to represent higher conservation value and provide higher land use stability for long-term conservation returns (NRMMC, 2009).

Category	Criterion	Variable	Datasets	Rationale
Vulnerability Prioritises smaller and more isolated patches that are less resilient and will benefit from increased connectivity	Habitat area	Total area of intact remnant vegetation within the PA (reflecting vegetation loss since the original PA data)	• NatureLink Perth PA data (O'Donnell, 2020) • Native vegetation extent (DPIRD, 2023)	Smaller habitat patches are less able to support viable populations and may need increased connectivity to other patches to meet minimum viable area requirements (Beninde *et al.*, 2015; Ramalho *et al.*, 2018; Peterson and Andrew, 2025). Total remnant vegetation was used instead of total patch area to more accurately estimate available habitat, assuming remnant components within each PA are functionally connected.
	Regional connectivity	Hanski's connectivity index, a functional landscape metric quantifying the connectivity of a patch based on the size and proximity of other patches in the network	• Derived from NatureLink Perth PA data (O'Donnell, 2020)	More isolated patches are less likely to retain stable metapopulations with surrounding patches, which can be improved through increased connectivity (Moilanen and Hanski, 1998; Hanski, 1999).
Ecological linkages				
Connectivity Finds linkages that will benefit from increased connectivity, but can realistically facilitate species movement and gene-flow	Linkage length	Length (m) of the linkage path	• NatureLink Perth ecological linkage data (O'Donnell, 2020)	Relates to the spatial extent of the intervention, and the likelihood of species dispersal as part of home range movements or intergenerational dispersal (Davis and Glick, 1978; Saura *et al.*, 2014). Longer linkages require rehabilitation over a larger scale with potentially less chance of successful gene flow, while very short linkages are more likely to be well-connected.
	Permeability	Average resistance per metre of the linkage	• Derived from NatureLink Perth ecological linkage data (O'Donnell, 2020)	Reflects how difficult it is for species to traverse the matrix based on underlying land cover and land use, which can help identify linkages that are already well-supported by habitat, or would be overly difficult to traverse even with rehabilitation interventions (Baum *et al.*, 2004; Kirk *et al.*, 2023).
	Maximum gap in habitat	Maximum distance between areas of remnant vegetation along the linkage, with distance calculated along the path	• Native vegetation extent (DPIRD, 2023) • NatureLink Perth ecological linkage data (O'Donnell, 2020)	Rehabilitation must be functionally connected to support animal dispersal (Baum *et al.*, 2004; Lynch, 2018). While remnant vegetation is not always habitat, it provides a relative measure of habitat continuity, helping to identify linkages that are either already functionally connected or too fragmented for meaningful improvement relative to species' dispersal capacity.

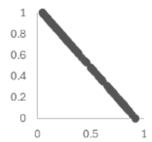
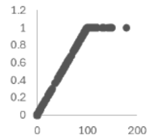
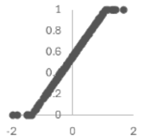
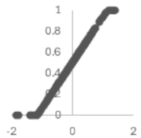
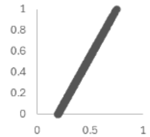
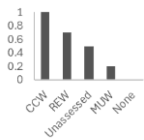
Category	Criterion	Variable	Datasets	Rationale
Route attributes Identifies opportunities and challenges to implementing an ecologically successful linkage	Barriers crossed	Number of barriers crossed by the linkage	• Derived from NatureLink Perth ecological linkage data and movement cost layer (O'Donnell, 2020)	Transport infrastructure, rivers and high-disturbance land uses can impede or prevent species dispersal between PAs (Forman and Alexander, 1998; Trombulak and Frissell, 2000; Shepard *et al.*, 2008).
	Surrounding vegetation	*Index of rehabilitation suitability* for surrounding vegetation, derived from the proportions of remnant, perennial, annual and unvegetated land covers within a 100m buffer of the linkage.	• Sentinel 2B satellite imagery: dates 6/08/2024 and 1/03/2025 (ESA, 2025) • Native vegetation extent (DPIRD, 2023) • Random forest-classified lawn data for the Perth region (Peterson and Andrew, 2025)	The types of vegetation (or lack thereof) surrounding the linkage can indicate the degree of habitat alteration and ecological novelty. This influences both the need for rehabilitation and the relative difficulty of re-establishing key habitat elements (Beninde *et al.*, 2015; Threlfall *et al.*, 2017; Klaus and Kiehl, 2021).

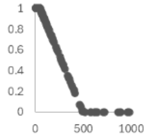
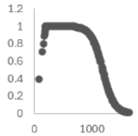
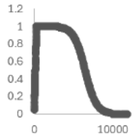
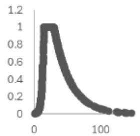
To ensure criteria can be meaningfully compared in MCDA, they need to be standardised to a common scale (Esmail and Geneletti, 2018). Criteria that characterised the linkages and PAs were therefore transformed to a 'criterion priority score' between 0 and 1 which represent how important the linkages and their relevant PAs are from a connectivity enhancement perspective. The relationships between the variables and the priority score were determined by consulting relevant literature, screening of priorities against local contexts, and iterative sensitivity testing. The priority scores for criteria in the *conservation value* category generally increased linearly with conservation value. Some variable transformations included a 'cap' or maximum value to reduce scaling effects from extreme values. For the categories of *vulnerability*, *connectivity* and *route attributes*, translations were more complex as they often represented unimodal or piecewise relationships (Table 7). For example, linkages with very small distances between remnant patches (<300m) are already likely to be highly functionally connected for a range of species, have less need for rehabilitation, and therefore received a lower priority score in the *linkage length* criterion. Conversely, longer distances between habitat are less likely to facilitate successful dispersal for many species even after rehabilitation, making functional connectivity less feasible (Saura *et al.*, 2014). Rationales for the calculation of priority scores are provided for each criterion in Table 7.

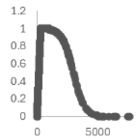
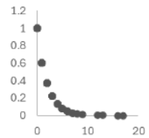
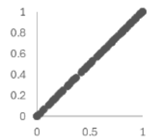
Once priority scores were calculated for all criteria, the scores characterising the PAs were translated to their associated linkages, as these were the final analysis units for the MCDA. For criteria in the *conservation values* category, each linkage was attributed with the mean score of its two respective PAs, which provided a balanced combination of the relative value of each. For criteria in the *vulnerability* category, the maximum priority score of the two PAs was taken to represent the linkage. This was to preserve the priority of linkages with at least one vulnerable PA, such as linkages that connect small and isolated PAs to large and secure ones.

To test for potential redundancies between criteria, pairwise correlations were also evaluated between the final priority scores of all criteria, using the Pearson correlation coefficient. Thresholds of -0.7 and 0.7 were applied to identify strong correlations (Dormann *et al.*, 2013), of which none were found between any of the criterion priority scores, suggesting no substantial overlapping influences or redundancies from individual criteria. Finally, the database of final priority scores for each individual criterion was linked to the spatial layer of the ecological linkages, to enable visualisation and interaction with the MCDA inputs.

Table 7. Details of the translation of each criterion to a standardised priority score; including a brief description, visualisation, and rationale of the relationships between the variables and criteria priority scores (x = variable, y = priority score). The descriptions specify the direction and type of transformation, as well as any normalisation, used to derive the standardised priority scores.

Category	Criterion	Relationship		Rationale
Conservation values	Vegetation complex rarity	Negative linear Min-max normalisation		Rarer vegetation complexes represent higher conservation value.
	TEC/PEC extent & diversity	Positive linear Min-max normalisation, capped at x ≥100		Higher TEC/PEC extent & diversity represents higher conservation value. Cap at score of 100 reduces scaling effects from highly diverse patches.
	T&P fauna richness (residuals)	Positive linear Winsorised normalisation, capped at the 2nd and 98th percentiles		Higher T&P species richness represents higher conservation value. Winsorisation reduces scaling effects of extreme values.
	T&P flora richness (residuals)	Positive linear Winsorised normalisation, capped at the 2nd and 98th percentiles		Higher T&P species richness represents higher conservation value. Winsorisation reduces scaling effects of extreme values.
	Formal protection	Positive linear Min-max normalisation		Higher proportion of formally PAs represents higher conservation value and long-term land use stability.
	Wetland values	Categorical CCW*: 1 REW*: 0.7 Unassessed: 0.5 MUW*: 0.2 None: 0		Priority scores of wetland management categories are based on the descriptions of relative ecological function/value, and ability to be restored, as per state government publications (DBCA, 2025d). A conservative priority value of 0.5 is applied to unassessed wetlands.

Category	Criterion	Relationship		Rationale
Vulnerability	Habitat area	Piecewise 0-50ha: priority = 1 50-500ha: linear decay ≥500ha: priority 0		Connectivity helps species meet minimum patch size requirements. Approximately 50ha is area required for urban-sensitive species (Beninde *et al.*, 2015). After 50ha, increased connectivity is likely to provide diminishing returns. With some uncertainty regarding specific patch size requirements and PA vegetation condition, 500ha was chosen as a conservative value at which the patch can fully support itself and connectivity improvements are likely to have very little effect.
	Regional connectivity	Piecewise 0-200: linear increase 200-700: priority = 1 700-max: sigmoidal decay		The Hanski's connectivity index used was relative rather than absolute (Rösch *et al.*, 2013). Because of this, thresholds were set based on iterative screening and sensitivity testing. The lower threshold acts as a buffer for extremely isolated patches with decreasing connectivity returns, while the upper threshold represents gradually decreasing need for rehabilitation where patches were already well-connected.
Connectivity	Linkage length	Piecewise 0-300m: linear increase 300m-3km: priority = 1 3-10km: sigmoidal decay		At 300m or less, the linkage drops below the dispersal thresholds of many species in the Perth region, even some terrestrial vertebrates (Watson *et al.*, 2001; Burbidge *et al.*, 2010; Nevill *et al.*, 2014; Ottewell *et al.*, 2019; Falla, 2022; How, 2024). There is less need for rehabilitation when many species can potentially traverse between habitats unassisted. Conversely, there is a drop in traversal feasibility with very long linkages. Very few species can disperse more than 5000m except for migratory birds and select other bird species (Rowley, 1983; Watson *et al.*, 2001; EPA, 2019; DAWE, 2022).
	Permeability	Piecewise 0-15: exponential increase 15-30: priority = 1 30-max: exponential decay		Linkages with very low average cost are likely to already be relatively easy for species to traverse, with native vegetation represented in the raster cost layer by values of 15 or less (O'Donnell, 2020). For the upper threshold, manual inspection of a wide range of linkages found those above an average cost of 30 were much more likely to pass through areas of high resistance which would present difficulties to functional connectivity improvements; while at an average cost of 50, linkages were likely to pass through multiple barriers or high-intensity urban areas. Priority flattens at average costs of 100 and above, which is the cost of major barriers in the raster cost layer (O'Donnell, 2020).

Category	Criterion	Relationship		Rationale
	Maximum gap in habitat	Piecewise 0-300m: linear increase 300m-1km: priority = 1 1-5km: sigmoidal decay, flattening to priority = 0 after ~4km		For the lower threshold, see the rationale for *linkage length* above. However, this variable adds nuance by considering the absolute distribution of existing habitat along linkages, regardless of length. With longer gaps between remnant vegetation, connectivity will be less feasible to restore as core habitats with crucial resources become few and far between (Saura *et al.*, 2014). A threshold of 1km (conservatively lower than that of 'linkage length') was set after a review of a range of Perth-based species dispersal (Watson *et al.*, 2001; Burbidge *et al.*, 2010; Nevill *et al.*, 2014; Ottewell *et al.*, 2019; Falla, 2022; How, 2024) to represent less feasible dispersal even with significant rehabilitation, thus indicating decreasing returns for effort. Priority reaches zero at a maximum gap of 5km (see rationale for *linkage length* above).
Route attributes	Barriers crossed	Exponential decay, reaching priority ≈ 0.1 at 5 barriers		Exponential decay was chosen to reflect the significant challenges that major barriers present to species movement (Forman and Alexander, 1998; Shepard *et al.*, 2008). Additional barriers have a diminishing impact on priority, as the probability of improving functional connectivity is already likely to be significantly hindered.
	Surrounding vegetation	Positive linear Min-max normalisation		Higher rehabilitation suitability of vegetation along the link represents a higher priority.

*Acronyms stand for management categories applied to wetlands by DBCA (2025d). In order from highest to lowest conservation value, these are:
CCW = conservation category wetland; REW = resource enhancement wetland; MUW = multiple use wetland.

3.2.2 Multi-criteria decision analysis

To assign each ecological linkage an overall priority score, MCDA was conducted using a weighted sum model. Weights were assigned to each criterion based on their relative importance to the overall rehabilitation priority. To do this, a top-down approach was used from core analysis units (PAs and linkages), to categories, and then criteria. The four categories were weighted first to allow deliberate balancing between these dimensions, ensuring that both PAs and linkages each received a total weight of 50. Relative weights were then assigned to criteria within each category to reflect their importance in relation to one another, before being scaled by their overall category to calculate absolute weights (Table 8).

Prior to assigning weights, the distribution of individual criteria priority scores and correlations between criteria were investigated to evaluate whether they provided useful variation or were introducing biases. Some adjustments to category and criteria weights were made based on criteria score distributions, however, no changes were made to reduce redundancy as there were no strong criteria correlations.

The *conservation values* category was given the highest weighting as these attributes embody conservation priorities, which is the primary aim of connectivity enhancement efforts. The category also had the largest number of independent criteria and provided the most variation in individual criteria priority scores. The *vulnerability* category was given a much lower rating as there were only two criteria that each showed relatively little variation in priority scores. The *connectivity* category was weighted slightly higher than *route attributes*, given it related more directly to functional connectivity of the whole linkage rather than opportunities or constraints along the route. Criteria weights were then allocated relative to their category. All *conservation values* and *vulnerability* criteria were assigned equal relative weights within their respective categories, except for formal protection, which was assigned half due to it being a land use designation that only indirectly relates to environmental quality. In the *connectivity* category, permeability was weighted slightly higher than linkage length. While length provided a proxy of dispersal difficulty due to distance, permeability provided a direct measure of how difficult that dispersal would be, and whether rehabilitation could realistically be achieved. Within *route attributes*, the barriers criterion was assigned a slightly higher weight than surrounding vegetation. This was to slightly reduce bias towards agricultural areas with lots of apparently 'suitable' areas for rehabilitation (i.e. annual vegetation), and to slightly increase the importance of linkages without major barriers, given their significant impedances to organism movement (Forman and Alexander, 1998; Shepard *et al.*, 2008).

Absolute weights were calculated by scaling the relative weights by their overall category weight. The overall priority score for each linkage was then calculated by multiplying the priority score of each criterion by their absolute weights and summing the weighted scores. With all absolute weights summing to 100, this gave each ecological linkage a possible scoring range of 0-100.

Table 8. Weighting system used to conduct the MCDA, showing each category weight, and individual criteria including relative weight within their category and their absolute weights which sum to 100.

Category		Weight	Criterion	Relative weight	Absolute weight
Protected areas	Conservation values	40	Vegetation complex rarity	1	7.27
			TEC/PEC extent and diversity	1	7.27
			T&P fauna richness	1	7.27
			T&P flora richness	1	7.27
			Formal protection	0.5	3.64
			Wetland values	1	7.27
	Vulnerability	10	Habitat area	1	5
			Regional connectivity	1	5
Ecological linkages	Connectivity	30	Linkage length	0.8	8
			Permeability	1.2	12
			Maximum gap in habitat	1	10
	Route attributes	20	Barriers crossed	1.2	12
			Surrounding vegetation	0.8	8
Total					100

3.2.3 Exploration of linkage priority scores

In addition to evaluating raw results, linkages were further analysed to understand spatial patterns in priority scores and drivers of priority across the region. First, linkages were flagged as either high or low priority if they scored within the top or bottom 10^{th} percentiles of linkage scores, respectively.

The study region was then divided into four development types – urban, peri-urban, rural, and natural – to evaluate patterns in linkage priorities across different development intensities. To classify development type, a 2km grid was established across the study area, removing areas of surface water (Geoscience Australia, 2024). Land cover percentages were calculated within each grid cell, using a decision tree framework to classify each cell into a development type (Figure 9). Land cover types were represented as presence/absence raster layers at a resolution of 10m. They included building cover (DPIRD, 2022), as an indication of development intensity; remnant vegetation (DPIRD, 2023), as a measure of relatively unaltered landscape; and annual vegetation and bare surfaces, which indicated the level of land clearing. Presence of annual vegetation and bare surfaces were calculated by comparing Sentinel 2B NDVI (normalised difference vegetation index, representing the density and greenness of vegetation; (Huang *et al.*, 2021)) imagery between the wet season (winter, 6^{th} August 2024) and dry season (immediately after summer, 1^{st} March 2025) (ESA, 2025). Annual vegetation was defined as pixels with NDVI > 0.2 in the wet season only, while bare surfaces had NDVI < 0.2 year-round.

Clear thresholds for development categories were not available in the literature, as vegetation and building cover varies widely with local urban form. Thresholds for urban and peri-urban areas were therefore determined through sensitivity testing and visual inspection of satellite imagery, informed by local knowledge. Tested values for building cover ranged from 5–30% for the lower urban threshold and 0.5–5% for the lower peri-urban threshold. A grid cell was automatically considered 'natural' if remnant vegetation cover was over 80%. For the edge-cases between natural and human-modified areas, land clearing and vegetation modification were inspected to determine whether an area was natural. Land clearing was assessed by the combined cover of annual vegetation and bare surfaces in the cell, while vegetation modification was assessed again by remnant vegetation. However, thresholds of vegetation clearing and modification were set at a lower 50% to represent the dominant land cover within the cell. Following sensitivity testing, the resulting development map was reviewed to ensure classifications aligned with on-ground conditions. Based on local knowledge, development types were considered appropriate and largely representative of the study area (Figure 10).

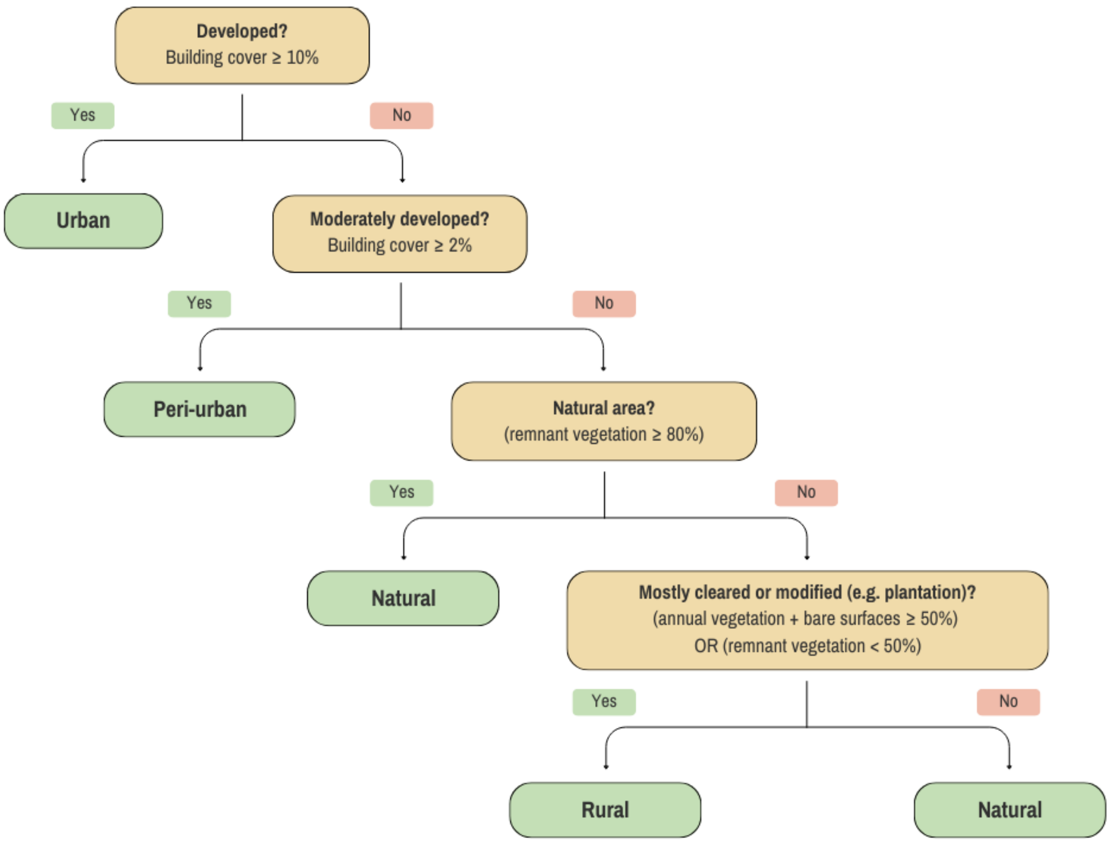


Figure 9. Decision tree used to classify development type based on building cover and vegetation characteristics. Classification thresholds distinguish urban, peri-urban, rural, and natural areas based on relative levels of built infrastructure, clearing, and vegetation intactness.Building cover and native vegetation data was sourced from DPIRD (2022, 2023); annual vegetation and bare surfaces data was calculated using Sentinel-2 NDVI satellite imagery (ESA, 2025).

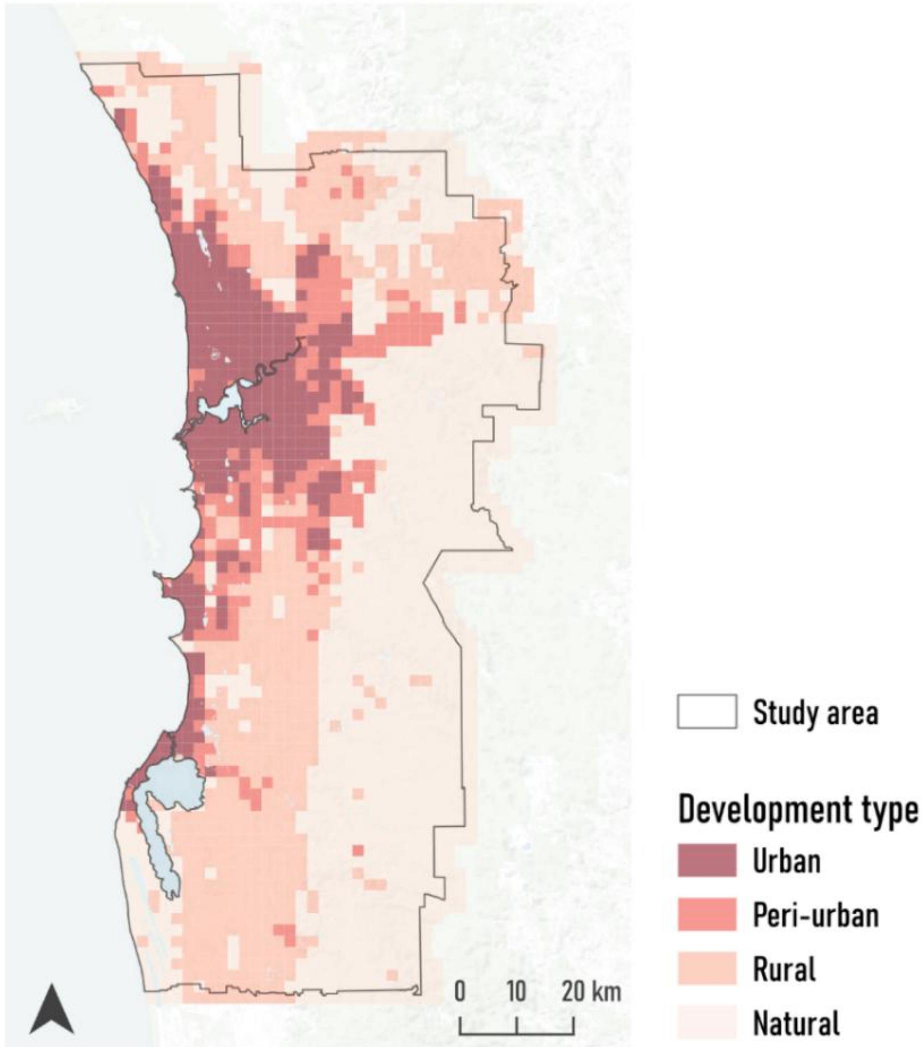


Figure 10. Map showing distribution of development types within the Perth-Peel region. Colours are organised from dark to light red, in order from highest to lowest development intensity.

To analyse differences in linkage priorities between development types, the two layers were intersected to calculate, for each linkage, its proportion within each development type. Linkages were assigned to their dominant development type, for those that predominantly (>80% of linkage length) occurred within one development type (940 linkages, 74.1%). The remaining linkages were assigned to a 'mixed' category for summaries but were not included in statistical analyses (below).

The number of high-priority and low-priority linkages in each development type were statistically compared using chi-square tests of independence. The mean linkage priorities and standard deviations were also compared across development types for five sets of data: the overall priority, and the priorities for each of the four categories (*conservation values*, *vulnerability*, *connectivity*, and *route attributes*). Mean scores were statistically compared within each set using one-way analysis of variance (ANOVA), with pairwise differences compared using tests of Tukey's Honest Significant Difference (HSD). Some of the priority scores (*vulnerability* and *connectivity*) were not normally distributed, which is an assumption of ANOVA. Additionally, Levene's test indicated unequal variances among development types ($p < 0.001$). However, given the large and reasonably balanced group sizes, ANOVA is generally robust to these violations (Schmider *et al.*, 2010). While the results remain informative, p-values were interpreted with appropriate caution.

To identify whether there were any criteria or categories driving priority across the region, a few additional analyses provided insight. Principal component analysis (PCA), in conjunction with permutational multivariate analysis of variance (PERMANOVA) and similarity percentages (SIMPER), were used to identify if any linkage characteristics were related to development types. Differences among development types were analysed using PERMANOVA based on a standardised Euclidian distance resemblance matrix, with SIMPER used to identify the primary criteria driving dissimilarities between groups. Additionally, Spearman's correlations were calculated between each category/criterion and the overall priority scores to identify which drove the most regional variation in priorities. Correlations were calculated for the whole region and within each development type.

3.3 Results

3.3.1 Spatial distribution of linkage priorities

Overall linkage priority scores ranged between 27.5 and 90.3, varying widely across the region (Figure 11). Because the landscape elements are relatively small compared to the study region, the spatial outputs are available online to enable closer examination on an interactive platform (https://arcg.is/11bKGe1).

High-priority linkages were most common south of Perth's city centre, especially in rural areas on the Swan Coastal Plain, as well as some clusters in peri-urban areas (Figure 11). Low-priority linkages predominated in natural areas, which characterised the eastern half of the study region in the Jarrah Forest (Figure 11). The number of low and high-priority linkages (i.e. 10^{th} and 90^{th} percentiles, respectively) varied significantly between development types, excluding the mixed category ($\chi^2 = 215.0$, df = 6, $p < 0.001$). The effect size was moderate (Cramer's V = 0.34), indicating that development type meaningfully influenced the distribution of priority classes. Rural areas had both the highest proportion of high-priority linkages (20.2%) and the lowest proportion of low-priority linkages (1.9%) within them. High-priority linkages were rarer within peri-urban and urban areas (11.3% and 3.2%, respectively, Figure 12), particularly so in urban areas where high-priority linkages were both sparse and more scattered (Figure 11). A total of 33.8% of linkages in natural areas were low priority, with only 0.5% being high-priority (Figure 12).

Comparing mean linkage scores across development types, similar results emerged (Figure 13). One-way ANOVA found significant mean differences between the four development types ($F_{3,935} = 134$, $p < 0.001$). Mean priority scores were highest in rural areas (71.2), and Tukey's HSD test revealed this was significantly higher than both urban ($M_{diff} = 5.5$, $p < 0.001$) and peri-urban (M_{diff} 5.7, $p < 0.001$) linkages. Natural linkages had substantially lower priority scores than linkages in the more human-modified landscapes (urban, peri-urban, and rural), trailing the next lowest (peri-urban) average by 14.6 points ($p < 0.001$). However, the mean difference between urban and peri-urban linkage priorities was not significant ($p = 0.99$,

Figure 13). While significant mean differences occurred between development types, the within-group variation was consistently high. Standard deviations ranged from 10.3 (urban) to 14.0 points (peri-urban), often outweighing between-group mean differences (Figure 13), indicating a broad range of priorities within each development type.

Variation in MCDA category scores also showed clear spatial patterns across the region. Conservation value tended to be higher on the coastal plain in the western half of the region (Figure 14), and in urban and rural development types rather than natural areas (Figure 15). The coastal plain also tended to have higher priority based on the connectivity and vulnerability categories (Figure 14), indicating a higher degree of fragmentation which would benefit from connectivity enhancement. Patterns were supported by one-way ANOVAs for each MCDA category, which all indicated significant differences among development types ($F_{3,935}$ = 134, $p < 0.001$ for all categories). Post-hoc Tukey tests showed that most of these differences were driven primarily by differences between altered landscapes and natural areas, which had consistently lower conservation values, connectivity, and vulnerability scores, resulting in separate Tukey groupings (Figure 15). However, vulnerability and route attributes categories drove additional differences in scores between development types. Urban and rural linkages had the highest vulnerability scores (Tukey group A), while peri-urban linkages and natural linkages were less vulnerable (group B). Route attributes scores most strongly differentiated the three altered development types (i.e. excluding natural). Priority scores in this category were highest on average in rural areas and lowest in urban areas (Figure 15), a result visible in the priority maps (Figure 14). The Tukey test found that urban route attributes were significantly lower on average than even natural linkages (M_{diff} = 4.4, p = 0.01), which consistently had the lowest priority scores in the other three categories (Figure 15). Together, these scores highlight why rural linkages tended to score higher in overall priority (Figure 13), with high vulnerability and more favourable route attributes. Conversely, urban linkages were highly vulnerable, but priorities were significantly constrained by the greater number of barriers and more challenging rehabilitation context typical of urban environments.

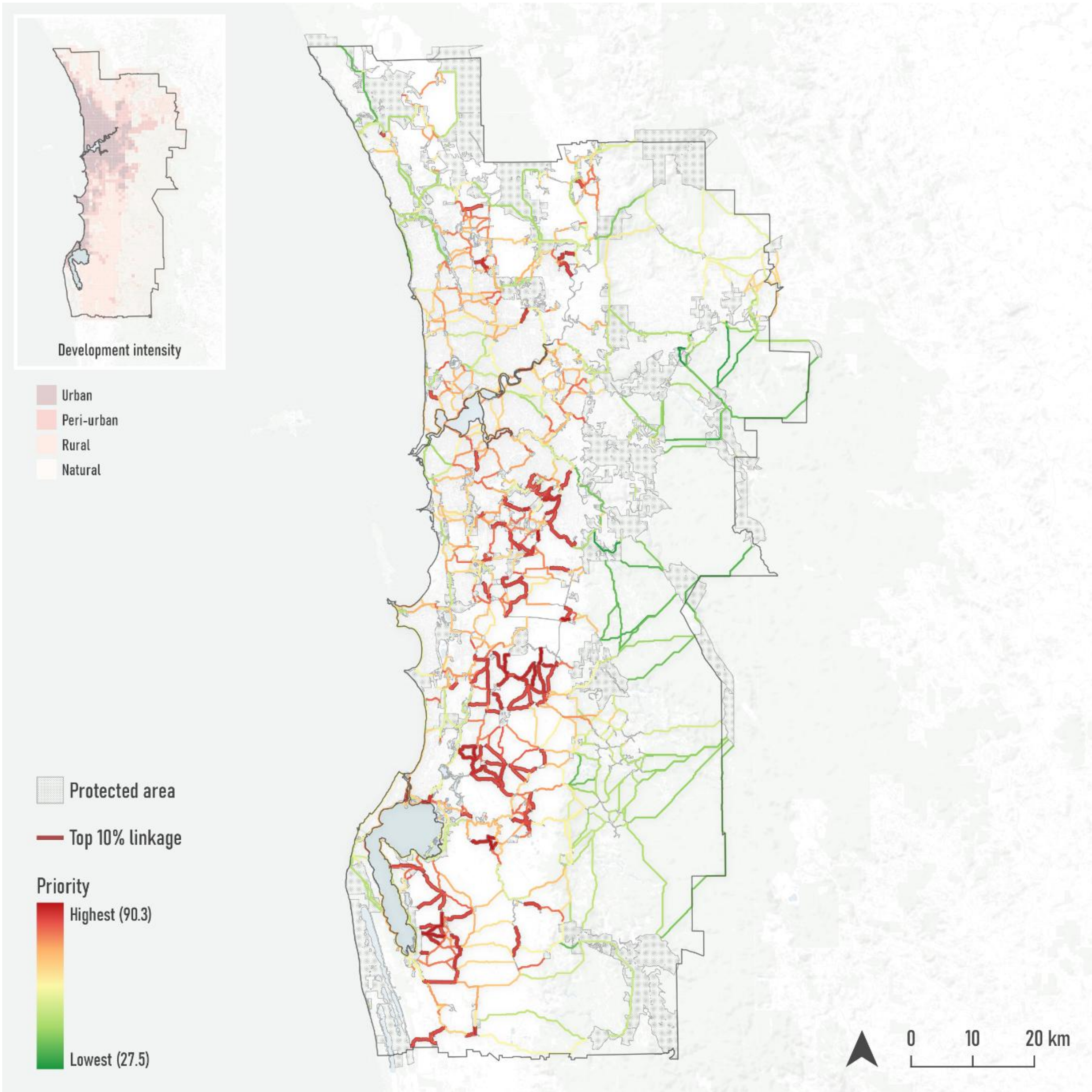
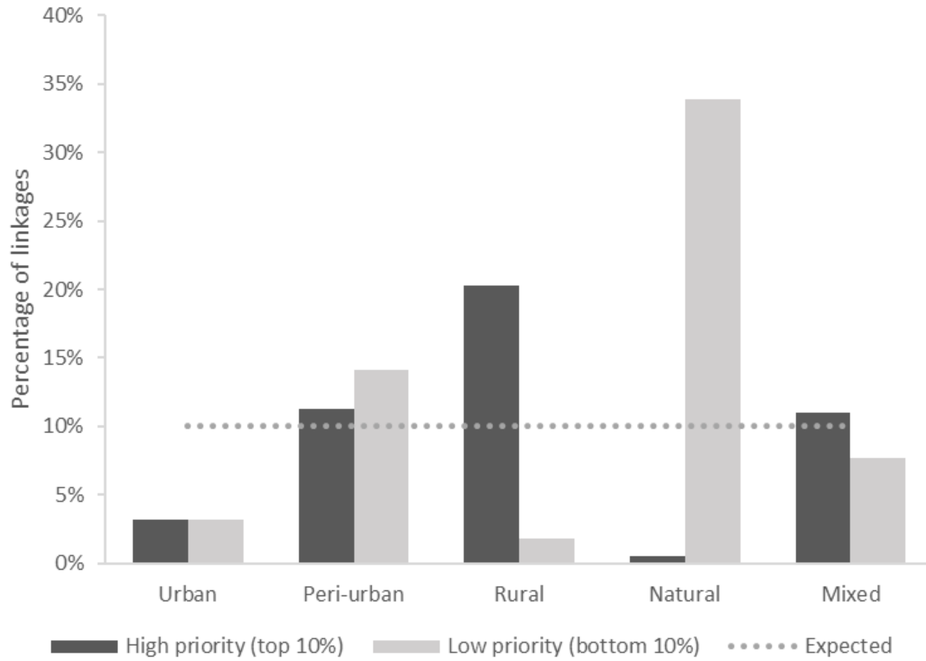
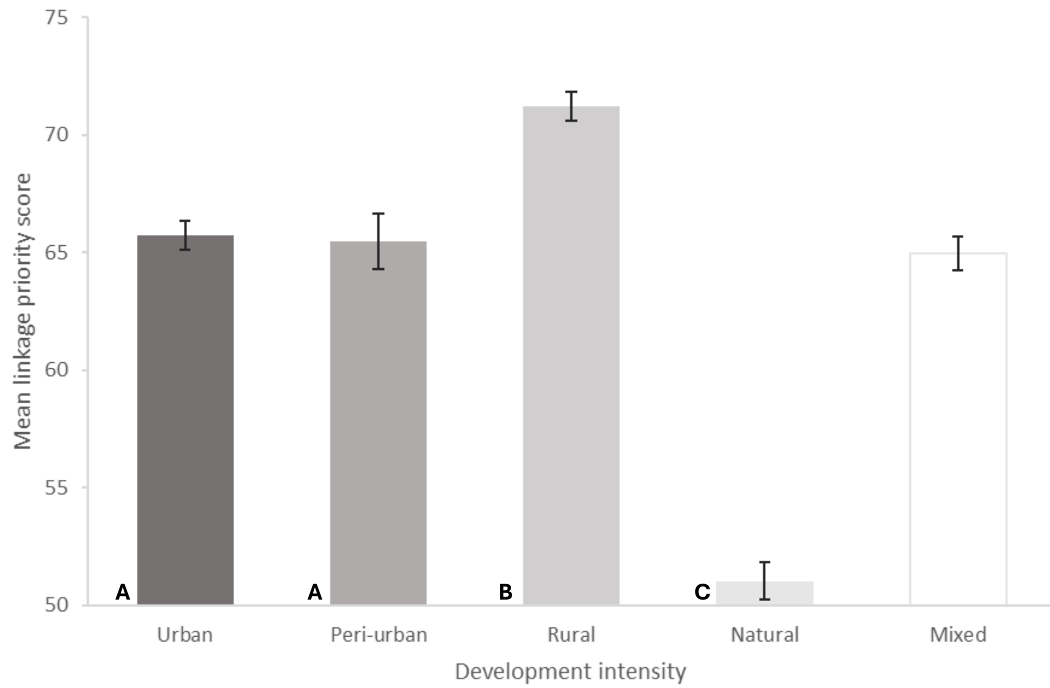


Figure 11. Spatial distribution of linkage rehabilitation priorities across 1268 linkages in the Perth-Peel region (black outline). Priority scores are represented by a colour gradient (see legend) stretched from lowest (27.5) to highest (90.3). High-priority linkages (top 10%) underlaid by thick red lines, with protected areas shown in grey hatching. Development intensities within the study area are indicated in the map insert (top left). Spatial outputs available here: https://arcg.is/11bKGe1.



Development type	Urban	Peri-urban	Rural	Natural	Mixed
Total linkage share	22.0%	11.2%	25.4%	15.7%	25.8%

Figure 12. Proportion of linkages identified as high-priority (black) and low-priority (grey) within each development type (arranged from high to low intensity). As high and low-priority linkages constitute the highest and lowest 10% of linkage scores by percentile, a 10% threshold line (light grey) shows the expected percentages across all linkages. The total linkage share of each development type is shown below the graph.



Development type	Urban	Peri-urban	Rural	Natural	Mixed
Mean priority score	65.74	65.50	71.22	51.04	64.96
Std. dev.	10.31	13.98	10.77	11.09	12.96
Std. error	0.62	1.17	0.60	0.79	0.79
Tukey grouping	A	A	B	C	-

Figure 13. Mean priority scores (including standard errors) of linkages across development types. Development types are arranged from high to low development intensity, with shading from dark (urban) to light (natural) and the 'mixed' category in outline only. The table below summarises the mean score, standard deviation, standard error, and Tukey groupings. Tukey letters (also shown on the graph above) indicate statistical similarity; types sharing a letter are not statistically different (α = 0.05).

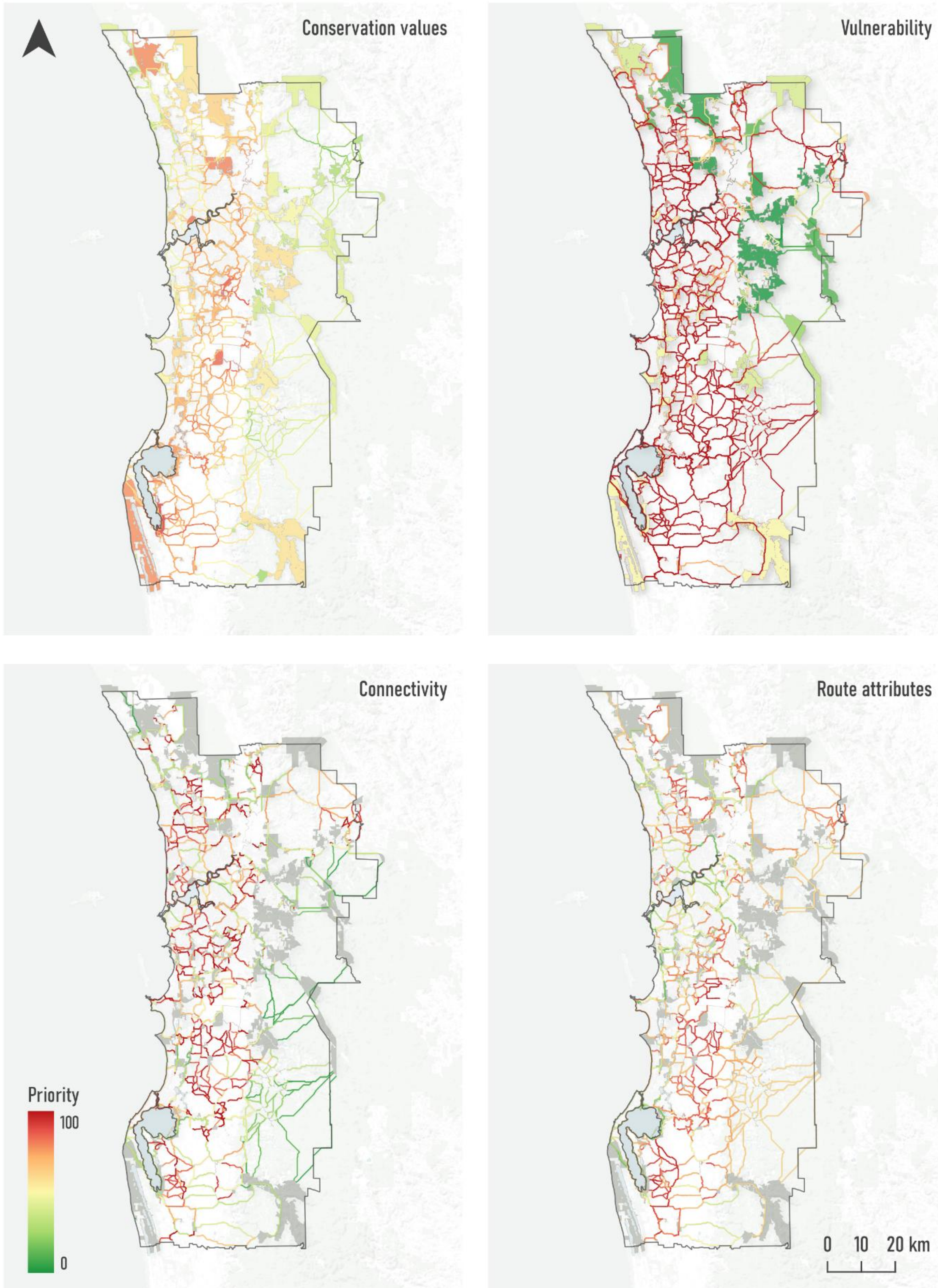
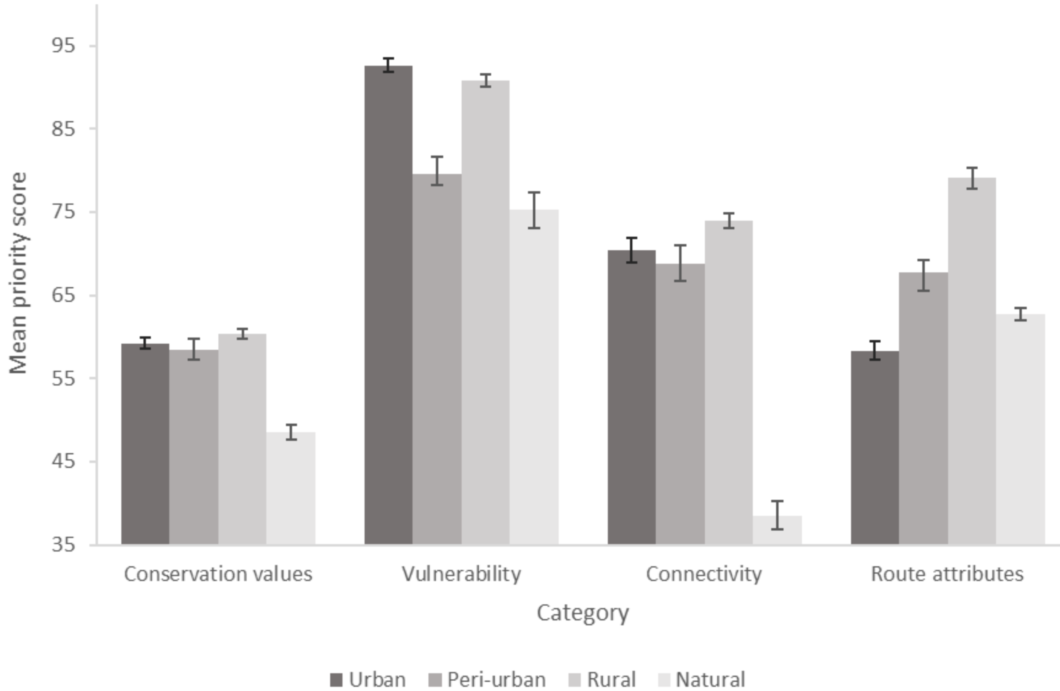


Figure 14. Spatial distribution of linkage rehabilitation priorities across Perth-Peel (black outline) within each category (see panel titles). Priority scores are represented by a colour gradient on a common scale from 0 to 100 (see legend) for both the linkages and the protected areas in the conservation values and vulnerability categories. For connectivity and route attributes, protected areas are shown in grey (scores are linkage-specific). Spatial outputs available here: https://arcg.is/11bKGe1.



Tukey groupings (α < 0.05)	Urban	Peri-urban	Rural	Natural
Conservation values	A	A	A	B
Vulnerability	A	B	A	B
Connectivity	A	A	A	B
Route attributes	A	B	C	D

Figure 15. Mean priority scores within each category (including standard errors) across development types. Categories are shown on the x-axis, while development types are arranged from high to low development intensity, with shading from dark to light. The table below summarises Tukey groupings within each category. Tukey letters indicate statistical similarity; types sharing a letter are not statistically different within that category (α = 0.05).

3.3.2 Drivers of linkage priority

It is of value to consider if there was any pattern to the way the thirteen criteria and four categories influenced prioritisation. Overall, there was an adequate distribution of priority scores across linkages providing useful separation of sites for rehabilitation (Figure 16). This was also true for the conservation values, connectivity and route attributes categories. However, vulnerability criteria were heavily skewed towards higher values (Figure 16, Figure 17) highlighting the high proportion of small, isolated protected areas in the region. The connectivity category exhibited a broad distribution with greater extremes and fewer average values, reflecting a more distinct separation of high- and low-priority linkages (Figure 16, Figure 17).

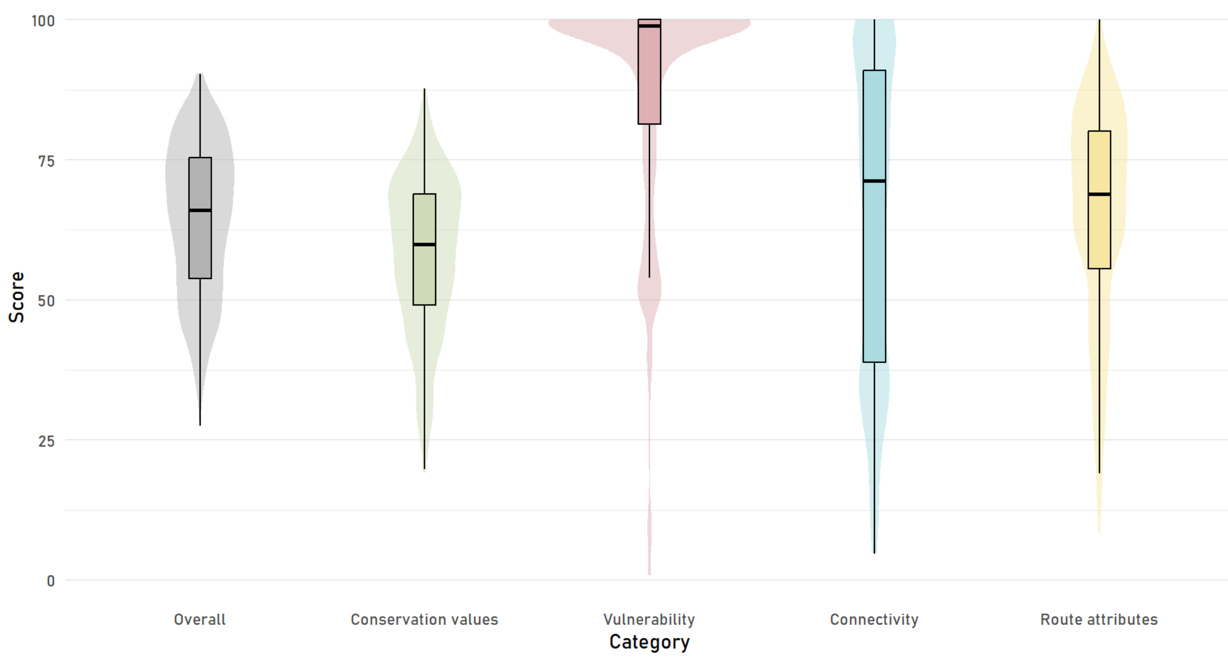


Figure 16. Violin plots showing distribution of linkage priority scores, overall (grey) and by category: conservation values (green), vulnerability (red), connectivity (blue), and route attributes (yellow).

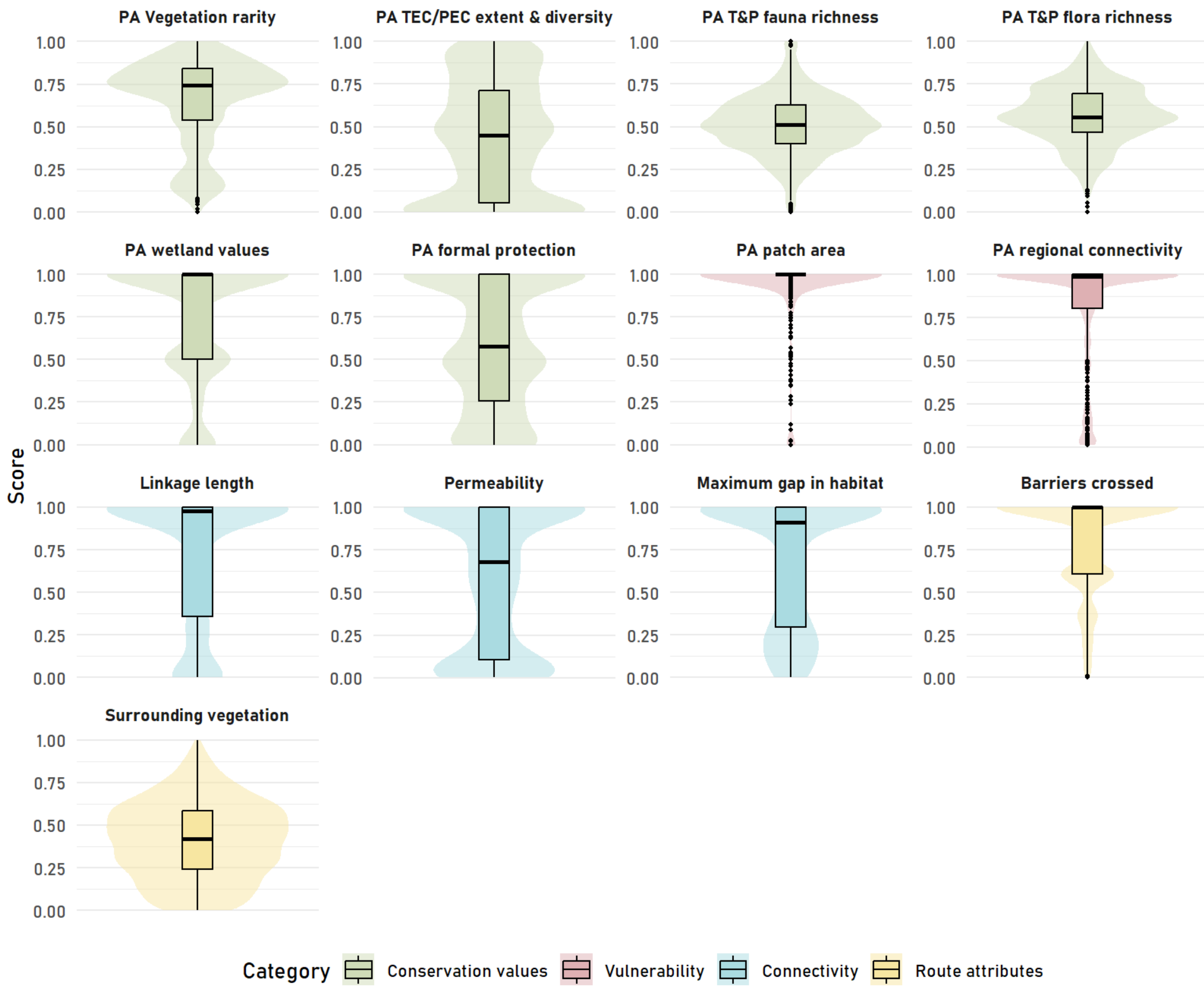


Figure 17. Violin plots showing distribution of linkage priority scores for individual criteria. Graphs are ordered and colour-coded by category: conservation values (green), vulnerability (red), connectivity (blue), and route attributes (yellow).

While there were some significant differences in linkage priority scores across development types, the drivers of rehabilitation priority were broadly unique to the individual linkage. Principal component analysis (PCA) investigated correlations in criteria priority scores across the four development types. Results revealed large within-group variability and between-group overlaps, especially within the three more human-modified development types (urban, peri-urban, and rural). Natural linkages, while also variable, occupied a distinct multivariate space relative to the human-modified types (Figure 18; Pseudo-$F_{3,935}$ = 56.22, $P < 0.001$). However, SIMPER results suggested this variation was primarily due to criteria differences between natural and more human-modified landscapes. Loadings show natural linkages tended to score lower on most of the criteria, including TEC/PEC extent and diversity, vegetation rarity, permeability, and the suitability of the surrounding landscape for revegetation (Figure 18). Most of the vegetation is intact in these areas, resulting in lower priority scores in these criteria.

High-priority linkages clustered in the lower right of the multivariate plot (Figure 18), where the priority scores of most of the criteria were highest. These linkages scored strongly on the PC1 gradient, driven by a synergy of connectivity and conservation-related criteria including *maximum gaps in habitat*, *permeability*, *vegetation rarity*, and *TEC/PEC extent & diversity*. They also positioned low on PC2 (though less strongly than PC1), which corresponded to higher levels of *formal protection*, *wetland values*, and *regional connectivity* (i.e. protected area isolation). Only a few criteria – notably *barriers* and *T&P flora* and *fauna* – were not aligned with this cluster, suggesting high-priority linkages did not consistently score strong in these dimensions overall. Barriers such as major roads were more common in developed areas which scored higher on many other criteria (Figure 18). As *barriers* had a negative exponential relationship with linkage priority, this criterion therefore in some instances downgraded linkage priority in contexts where most other criteria increased it.

The lower right region of the PCA also showed mild clustering of rural linkages, consistent with the higher number of high-priority linkages found in this development type. However, the effect size reported in the PERMANOVA test of differences between development types was relatively small ($R^2 = 0.15$), highlighting the mildness of this clustering and leaving much of the variability in priority scores to be explained. This small effect size and large visual overlap within the PCA results confirm that the characteristics of individual linkages were highly variable within development types, primarily shaped by their specific contexts.

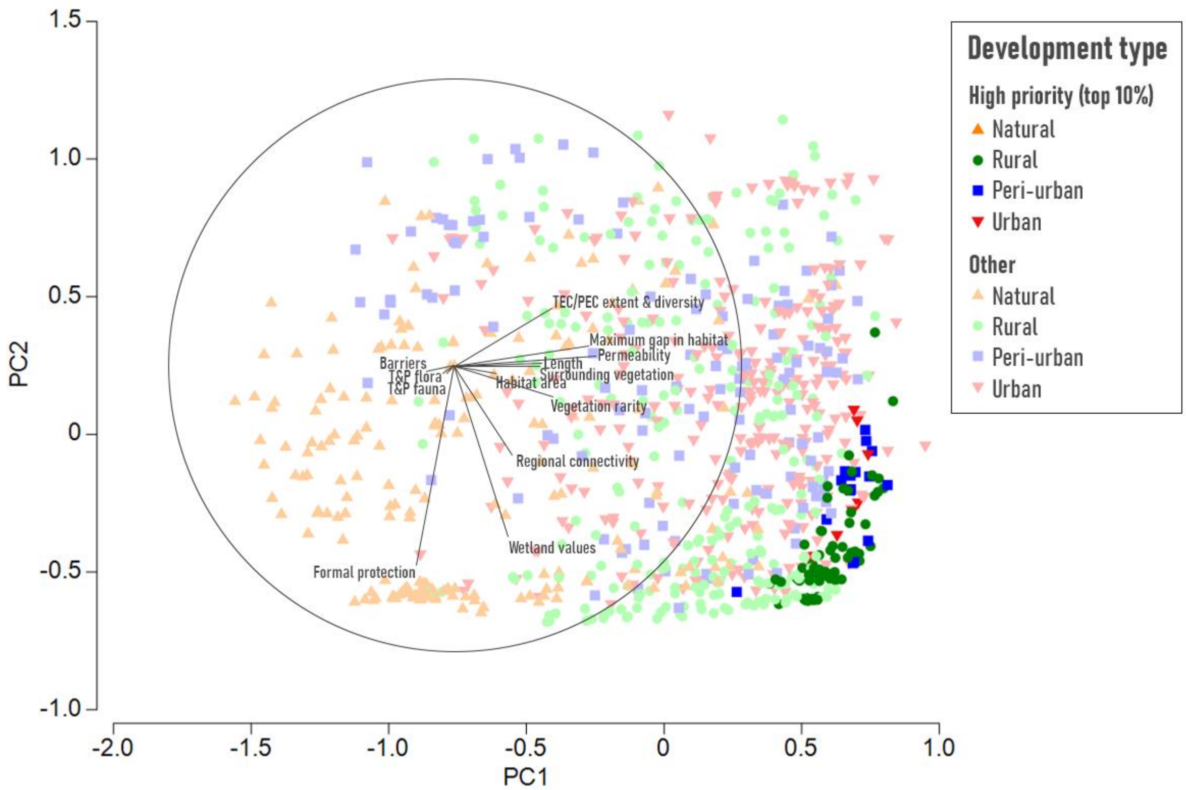


Figure 18. Principal Component Analysis (PCA) of individual linkage criteria scores, categorised by development type. High-priority linkages are shown in full colour, while other linkages are faded. Points are ordinated based on Euclidean distance. PC1 and PC2 explained 24.5% and 13.4% of the total variation, respectively. Black vectors represent loadings, indicating the direction and strength of each criterion's contribution to the ordination. (PERMANOVA: Pseudo-F = 56.22, $P < 0.001$, $R^2 = 0.15$).

Although conservation values were weighted more heavily in the MCDA (40%), connectivity (30%) emerged consistently as the most influential driver of variation in linkage priorities across the region, both overall and across all development types (Figure 19A). This reflects the greater spread and more extreme values of connectivity criteria (Figure 17), which, combined with the smaller number of criteria within this category, increases their relative influence and causes less within-category dampening of extremes. This highlights the greater multi-dimensionality of conservation value priorities across the region, while connectivity priorities were generally more aligned and discrete. Consequently, connectivity varied more strongly across linkages (Figure 16) and therefore contributed more strongly to differentiating priorities.

The effect of connectivity on driving priority scores was most pronounced, especially relative to other categories, in the urban environment (rho = 0.86) (Figure 19A). Linkage length and permeability, two criteria from the connectivity category, were the two most influential criteria in this development type (Figure 19B). Conservation values and vulnerability had the least effect in the urban environment compared to the other development types – likely related to widespread clearing and resulting rarity that increases the conservation value of all remaining reserves. In contrast, the influence of conservation values was greater in the less-developed landscapes (peri-urban, rural) (Figure 19A). There was slightly higher variation in conservation values scores (SD_{urban} = 11.8, compared to $SD_{peri\text{-}urban}$ = 13.5 and SD_{rural} = 13.7).

While vulnerability was consistently the least influential category on overall priorities, its influence noticeably changed across development types. In less-developed contexts, vulnerability contributed more strongly to variation in overall priority scores (Figure 19A). Vulnerability criteria were heavily skewed towards high priority, with a long tail of lower-vulnerability linkages (Figure 16, Figure 17). Linkages in more-developed landscapes were more consistently vulnerable (Figure 14), forming the bulk of high-priority linkages, while the broader variability in priorities was more present in less-developed landscapes.

Among the 13 criteria, those that drove the most variability in rehabilitation priority played a strong role in describing the level of landscape modification around the linkage. Permeability drove most variation across the region (rho = 0.68) (Figure 19B). With its broad distribution (Figure 17), it strongly distinguished low-priority linkages (high permeability, mostly vegetated) and high-priority linkages (low-moderate permeability, in cleared or altered landscapes). This pattern reflected a strong contrast between the largely vegetated Jarrah Forest and the more fragmented coastal plain – a pattern which contributed to much of the variation in overall priority across the region (Figure 11). Surrounding vegetation suitability for rehabilitation (rho = 0.65) had a narrower distribution but similarly described how 'intact' a linkage was through its measure of native vegetation along the route (Figure 17). This criterion also correlated moderately with permeability (Figure 20). Rarity of vegetation complexes (rho = 0.57) and maximum gap in habitat (rho = 0.50) further reinforced this alignment of priority with linkage modification, distinguishing the fragmented western linkages (rarer vegetation types, larger gaps) from intact eastern linkages (common vegetation types, smaller gaps) not needing rehabilitation. However, rather than simply rewarding fragmentation, these criteria also penalised priorities in extremely fragmented areas that could present overwhelming connectivity or rehabilitation obstacles. Some of the strongest drivers – permeability, surrounding vegetation suitability, maximum gap in habitat, and linkage length – limited priorities of linkages that were too impermeable, fragmented, or difficult to rehabilitate. They essentially balanced the need for rehabilitation with its feasibility. Overall, these criteria show that overall rehabilitation priorities were largely driven by patterns in landscape modification rather than protected area characteristics.

Threatened and priority (T&P) flora and fauna richness drove the least variation in priority scores across the region (Figure 19B). Their priority scores provided less variation than most other criteria as they were more narrowly distributed (Figure 17), and were also largely uncorrelated with other criteria (Figure 20). While the two criteria were weighted at a combined 14.54% to reflect their ecological importance, the PCA (Figure 18) and correlation matrix (Figure 20) indicate that they provided unique ecological information that was less aligned with the dominant spatial gradients that shaped other criteria. As a result, their influence was not reinforced through the multivariate structure of the MCDA and contributed little to variation in overall priority across the region. However, this does not mean they didn't influence priorities at the scale of the individual linkage – rather, priorities across the region were not driven by patterns in these criteria.

Other criteria in the conservation values category were more moderately correlated to overall priority scores (Figure 19B), suggesting that while they did meaningfully shape regional patterns in rehabilitation priority, they were not as sharply discerning as the criteria describing fragmentation and development across the region.

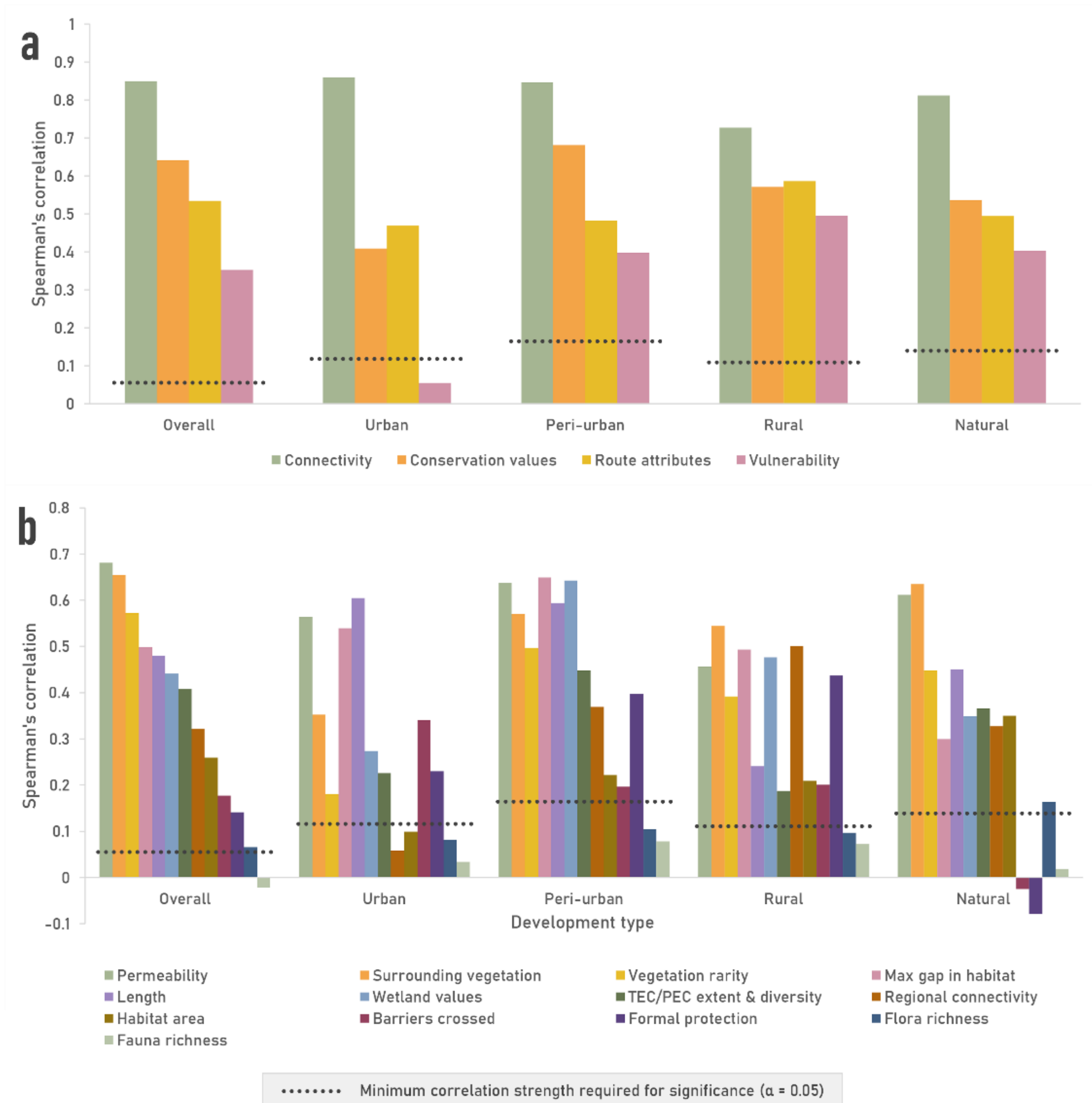


Figure 19. Spearman correlations between overall priority scores and a) category score, and b) individual criteria scores, calculated across all linkages (overall) and within each development type (urban, peri-urban, rural, natural). Bars are ordered by overall correlation strength. Dotted lines indicate the minimum correlation required for significance (α = 0.05), based on sample size. Bar colours match legend items, which follow the bar order (standard reading order, left to right).

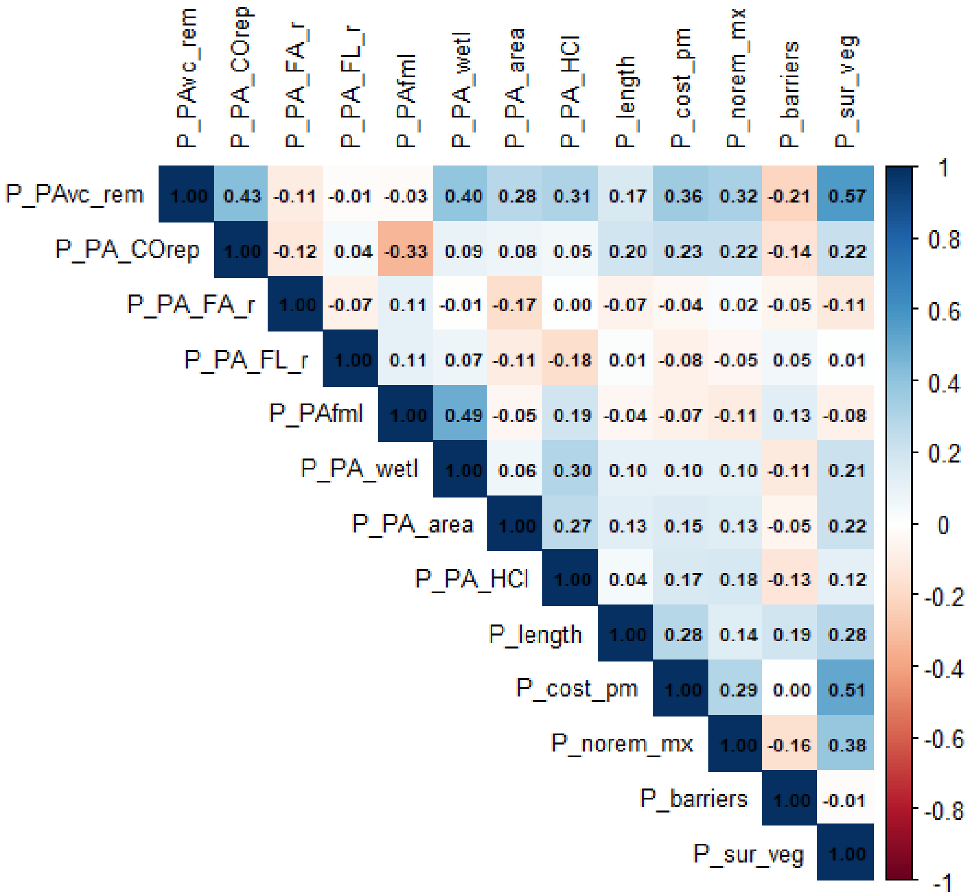


Figure 20. Spearman correlation matrix showing relationship strength between criterion priority scores across all linkages. Colours indicate correlation strength: dark blue = positive, white = none, dark red = negative. Significance threshold: ρ ≈ 0.06 (n = 1268, p <0.05).See below for criteria code definitions:

P_PAvc_rem = PA vegetation rarity
P_PA_COrep = PA TEC/PEC extent & diversity
P_PA_FA_r = PA threatened fauna richness
P_PA_FL_r = PA threatened flora richness
P_PAfml = PA formal protection
P_PA_wetl = PA wetland values
P_PA_area = PA habitat area
P_PA_HCI = PA regional connectivity
P_length = Linkage length
P_cost_pm = Permeability
P_norem_mx = Maximum gap in habitat
P_barriers = Barriers crossed
P_sur_veg = Surrounding vegetation suitability

3.4 Discussion

3.4.1 Balancing conservation value and landscape feasibility

Regional linkage mapping is a valuable conservation and connectivity planning tool, producing outputs that are applicable to a wide range of implementation outcomes, from conservation and protection to habitat enhancement and land use management (O'Donnell, 2020). With limited resources and a wide range of protected areas and linkages to support, it can be difficult to translate linkages into spatially targeted, real-world interventions. Using MCDA to view linkages through the lens of a specific goal, this study guides rehabilitation efforts by identifying which linkages may support the most efficient connectivity gains to areas with the most need.

Without access to connectivity planning tools, it can be tempting to target efforts based on the presence of species or communities of conservation concern (Newell *et al.*, 2025). This is especially so in the Perth-Peel region of WA, in which rich and threatened biodiversity and state legislative structures create a strong focus on conservation significance in environmental management (DBCA, 1986, 2016; Hopper and Gioia, 2004). However, potential connectivity gains are not dictated by conservation value alone – they are also strongly linked to underlying landscape patterns. In fact, landscape structure – degree of modification and fragmentation – was a primary factor driving linkage rehabilitation priorities across the region. This is not to diminish conservation values, as there are many rare and threatened urban species that require targeted intervention (Ives *et al.*, 2016; Soanes and Lentini, 2019). Instead, it highlights an important tension in connectivity planning: there are diminishing connectivity returns from rehabilitation if the landscape is already intact, and there are unlikely to be returns without intensive effort if the landscape is too extremely fragmented for local species (Oliveira-Junior *et al.*, 2020). This highlights the importance of prioritising rehabilitation based not only on the ecological value of the areas to be connected, but on how necessary and feasible both rehabilitation and functional connectivity are from a landscape perspective. Acknowledging this limitation allows us to realise the greatest conservation outcomes while avoiding potential 'empty pursuits' with limited ecological return on investment.

3.4.2 Connectivity opportunities in rural and urban areas

Rural landscapes demonstrated the greatest opportunities for connectivity enhancement, with a disproportionate number of high-priority linkages and more suitable route attributes for rehabilitation than other development types. Rural landscapes tended to have less barriers and be covered by land cover that may be feasible to rehabilitate, sometimes offering patches of native vegetation on which rehabilitation could be anchored. The surrounding vegetation criterion, which was a strong driver of these patterns, did introduce some complications. Priorities for this criterion were based on rehabilitation need and feasibility, meaning remnant vegetation and bare areas were downgraded as having low need and feasibility, respectively, while perennial and annual vegetation types were upgraded. This approach may have undervalued linkages with degraded remnant vegetation and inadvertently elevated widely cleared agricultural areas. However, this criterion carried a relatively low weighting, and degraded remnant patches were relatively rare in the fragmented coastal plain compared to cleared areas. It is therefore unlikely to introduce strong biases and instead reflects the reduced ecological complexity and increased feasibility of rehabilitation in rural areas. Though not part of this study, rehabilitation may also be less constrained by socio-economic factors in rural areas, where land ownership and land-use competition are typically less complex than in urban environments (Ingram, 2008; Aronson *et al.*, 2017). Together, these factors suggest that rural linkages may offer high-leverage opportunities for achieving meaningful connectivity gains.

Inversely, the effects and limitations of landscape structure exerted much more relative influence in urban linkages than in other development types. High-priority linkages were relatively rare, and overall priorities were considerably aligned with linkage permeability, length, maximum gaps in habitat, and presence of barriers. This pattern aligns with extensive literature on the challenges to urban connectivity and restoration (Davis and Glick, 1978; Ingram, 2008; Shepard *et al.*, 2008; Klaus and Kiehl, 2021), also evident in Perth

(Stenhouse, 2004). Vulnerability criteria did not significantly influence priorities in urban areas, and while conservation values showed variation, they also generally had much less relative influence in this development type. Criteria with greater variation and stronger correlations between them – primarily the physical and structural attributes of linkages – became the main determinants of overall priority. Many of Perth's protected areas are already vulnerable and contain biodiversity under threat (How and Dell, 2000; Ramalho *et al.*, 2014), so it is understandable that those linkages with greater physical potential to support functional connectivity for a range of species will unlock the benefits of enhanced connectivity. With high competition over land use and a need to balance myriad human and ecological requirements (Aronson *et al.*, 2017; GANSW, 2024), urban linkages which offer these advantages become disproportionately important for enhancing and maintaining network connectivity, especially where they connect high biodiversity values. High-priority urban linkages are rare – making the impetus for action greater than in any other development type.

3.4.3 Linkage-specific connectivity planning

While generalisations were made about rehabilitation priorities in various development types, there is also a need for linkage-specific planning. The high variability of linkages within development types, supported by PCA analysis, suggests that development type alone is insufficient to describe rehabilitation priorities – instead, they are shaped mostly by their individual contexts. Given the high spatial heterogeneity of altered human environments, particularly urban ones (Cadenasso *et al.*, 2007), this is perhaps not surprising, but it is nonetheless important. Broad spatial generalisations about rehabilitation priorities are unlikely to produce optimal conservation outcomes in any case, let alone in complex environments with many interacting factors shaping the suitability of each option. This highlights the usefulness of spatially explicit, multi-criteria-based tools in urban connectivity and rehabilitation planning. With many competing and overlapping factors influencing the priority of a linkage, the tool offers structured and transparent decision-making, while accommodating for environmental independence across linkages.

The research aimed to answer questions relating to connectivity enhancement. However, ecological connectivity planning involves a range of management options, for example conservation, greenfield development planning, or identification of critical patches for restoration (Qi *et al.*, 2021; Cameron *et al.*, 2022). This study does not identify the most currently 'valuable' linkages, but instead identifies which may experience the greatest potential benefits from rehabilitation based on the ecological need for connectivity and the feasibility of enhancing it. To answer alternative questions, such as identifying and conserving linkages of high existing connectivity value, MCDA approaches may prove again useful, specifically for the ecological network in this study.

3.4.4 Limitations and use considerations

The prioritisation methodology used in this study focused primarily on ecological need and feasibility. While environmental and socio-economic factors sometimes overlap, for example when greater rehabilitation feasibility translates to lower costs (Ingram, 2008), the focus was primarily on ecological considerations. This approach may miss considerations of socio-economic factors that can be crucial for the success of urban rehabilitation (Klaus and Kiehl 2021). In fact, uncertainties surrounding the preferences or capabilities of land managers and conservationists became apparent in defining the relationships between raw variables and rehabilitation priority. For example, ideal linkage lengths and maximum gaps between habitat were informed by wildlife dispersal thresholds, but necessary assumptions were made around what made these 'too much' in terms of effort. Furthermore, with scarce and piecemeal research on the movement of local species, threshold-setting was based on limited information. These uncertainties highlight the potential value of participatory MCDA in future research. By consulting planners and ecologists in criteria identification, threshold setting and weighting, MCDA could help to supplement these uncertainties with local knowledge (Hage *et al.*, 2010; Sutherland and Burgman, 2015). This addresses both data limitations and the underlying socio-ecological system where ecological and management feasibilities are intertwined (Aronson *et al.*, 2017).

In addition, the analysis integrates diverse ecological datasets and modelling approaches, each carrying inherent limitations that reflect the best available information rather than perfect representations of on-ground conditions. Variability in data age, spatial resolution, survey effort, and verification – particularly for threatened species or communities records, and wetland values – introduce some uncertainty into the underlying layers. Methodological assumptions, including in regional connectivity calculations, cost layer simplifications, or definition of 'suitable' vegetation for rehabilitation, also introduce potential bias or noise into the prioritisation. However, these limitations were minor and mitigated through data filtering, statistical bias correction and conservative scoring (Appendix A), and are not expected to substantially alter the overall spatial patterns identified. Nonetheless, future research would benefit from updated data and refined methodologies to further reduce uncertainties.

3.4.5 Conclusion: guiding strategic regional linkage rehabilitation in Perth-Peel

Regional rehabilitation priorities for connectivity in the Perth-Peel region are not driven by site-level ecological values alone – rather, they are driven by patterns in landscape fragmentation and development. We must acknowledge that to achieve optimal biodiversity outcomes from enhanced connectivity, rehabilitation must target landscapes that have both need and feasibility (Klaus and Kiehl, 2021). This is particularly true for inner city areas – intense development can present challenges for functional connectivity and rehabilitation (Ingram, 2008; Ahern, 2013; Lynch, 2018) – meaning feasible linkages are important for conservation. In contrast, rural areas on the Swan Coastal Plain offer greater opportunities and potentially substantial conservation outcomes from targeted intervention. Despite these broad patterns, however, linkage-specific planning remains essential in highly heterogenous human environments. While future work could integrate socio-ecological considerations through participatory approaches, this MCDA framework offers a clear foundation for strategic connectivity planning across the Perth-Peel region.

CHAPTER 4:

IDENTIFYING LOCAL REHABILITATION PRIORITIES FROM CIRCUITSCAPE CONNECTIVITY MODELLING

4.1 Introduction

4.1.1 Context and aims

Ecological linkages – made up of vegetated corridors, stepping stones, and areas of permeable landscape matrix – are essential to facilitate dispersal between isolated core habitats or reserves. However, for a linkage to be effective and support biodiversity into the future, it needs to be functionally connected for a range of species (Saura *et al.*, 2014; Lynch, 2018). The degree of functionality is inherently species-specific, dependent on their habitat requirements, movement abilities, and tolerance to urbanisation (Keeley *et al.*, 2021). If a linkage only supports a narrow range of species, few gain access to the benefits of connectivity while most biodiversity remains isolated. In contrast, linkages that support the dispersal of multiple wildlife guilds can foster complex and diverse ecological interactions in habitats on either side of the linkage, which is the foundation for ecosystem resilience (Peterson *et al.*, 1998).

Because movement is tied to the organism, functional connectivity often operates at local scales and is shaped by local landscape features. Wildlife may follow resources such as food or shelter (Ottewell *et al.*, 2019; Rycken, 2019), while threats and barriers can reduce or redirect movement within the matrix (Bryant *et al.*, 2017; Ferronato *et al.*, 2024; Santoro *et al.*, 2024). Wildlife are also constrained by dispersal limitations in the Perth-Peel region where dispersal thresholds can range from kilometres to hundreds or even tens of metres (Russell and Rowley, 1993; How and Dell, 1994, 2000; Bradshaw and Bradshaw, 2002; Rycken, 2019). These often-short dispersal thresholds mean that even modest gaps in habitat can disrupt movement. While regional linkages identify broad movement paths, they do not capture where – along the linkage or in the surrounding landscape – this connectivity is weak or severed, which is essential for identifying on-ground rehabilitation priorities. Therefore, understanding the functional connectivity of multiple wildlife guilds at a local scale is essential to ensure rehabilitation can realistically contribute to dispersal (FitzGibbon *et al.*, 2007; Fletcher *et al.*, 2016), potentially multiplying connectivity benefits with targeted interventions (Croeser *et al.*, 2024).

Identifying on-ground rehabilitation priorities also requires understanding the complex nature of organism movement through a landscape. Least-cost paths identify a single optimal route between habitats, assuming organisms have complete prior knowledge of the landscape it traverses (McRae *et al.*, 2008). This can lead practitioners to overlook alternative routes or underestimate implementation flexibility when identifying potential rehabilitation sites (Pinto and Keitt, 2009). In reality, wildlife have access to multiple dispersal routes and there is considerable uncertainty involved in predicting these movements. To better reflect these dynamics, circuit theory offers a complementary approach (McRae *et al.*, 2008). By modelling the landscape as an electrical circuit – a series of resistors – this approach allows movement to be simulated as electrical current flows across the landscape. It sees organisms as 'random walkers', acknowledging their limited knowledge of the landscape and ability to choose from potential pathways (McRae *et al.*, 2008). Outputs describe movement probability along a gradient across the landscape, which can provide practitioners with detailed information on functional connectivity to inform rehabilitation priorities both along and around the linkage. Circuit theory has been used globally in various contexts to inform ecological network planning (Dickson *et al.*, 2019), and there has been a recent surge of studies using the tool to prioritise areas for restoration (e.g Cameron *et al.*, 2022; Liang *et al.*, 2023; Molné *et al.*, 2023; Ortega *et al.*, 2023; Wu *et al.*, 2023). However, these studies have generally focused on the conservation and restoration of high-movement ecological corridors or barrier detection, rather than on where to rehabilitate.

This research aims to develop a local-scale ecological connectivity framework that identifies rehabilitation priorities using functional connectivity models incorporating dispersal limitations for multiple wildlife guilds. The objectives are to:

1. Quantify baseline functional connectivity using Circuitscape for four wildlife guilds within three case study sites in contrasting landscape contexts.
2. Derive spatially explicit rehabilitation priorities from model outputs using a consistent and transparent framework, providing guidance to optimise connectivity between protected areas.

4.1.2 Case study sites

Three case study sites were selected to develop and demonstrate the use of the framework. The sites contained select high-priority ecological linkages identified in Chapter 3, representing a range of development contexts – existing urban, emerging urban, and rural – to investigate their unique influence on rehabilitation plans (Table 9). While the *existing urban* and *rural* sites mainly correspond within the urban and rural development types introduced in Chapter 3 (Figure 10), the *emerging urban* site crosses a mix of urban and peri-urban landscapes, where new suburbs are actively being developed.

Table 9. Summary of the high-priority linkages from Chapter 3 selected for the local case studies. From left to right, the table shows which site the linkage belongs to, the linkage ID, its overall priority score, and its individual category scores (conservation values, vulnerability, connectivity, and route attributes).

Site	Linkage ID	Overall	Conservation values	Vulnerability	Connectivity	Route attributes
Existing urban	*563*	81.5	79.1	100	97.3	59.1
Emerging urban	*654*	87.1	83.4	97.1	97.5	73.9
	624	82.7	72.5	97.1	98.5	72.2
Rural	*1194*	83.2	72.0	100	85.5	93.8
	1197	84.1	79.5	100	81.9	90.8
	1210	88.8	83.7	100	86.3	97.0

Existing urban

The existing urban site is located approximately 10km south of Perth's city centre in an established suburban context (Figure 21). It covers approximately 15km^2 and contains one ecological linkage which joins two protected areas to the north and the south of the site, both part of Beeliar Regional Park. The linkage had a priority score of 81.5, scoring highly in conservation values, vulnerability and connectivity, but more modestly in route attributes, reflecting a potentially difficult rehabilitation context (Table 9).

The site is predominantly residential in the north, with scattered parks and schools, while the south of the site contains adjacent educational and medical campuses (Figure 21). Perth's largest freeway and train line runs along a north-south axis in the east, and an arterial highway runs from west-east across the centre of the site, presenting major movement barriers especially to ground-based fauna. Much development has already occurred, with peak residential development occurring in the 1970s and 1980s. Remnant vegetation is mostly associated with the protected areas and the university or medical campuses. Limited native vegetation occurs outside these areas, except for within scattered parks and narrow strips along roads, typically consisting of banksia woodlands with melaleuca woodlands in damper areas (DBCA, 2018b). However, both protected areas contain conservation category wetlands (DBCA, 2024) and native vegetation in the area supports a range of endemic fauna, including quenda (*Isoodon fusciventer*), southwest snake-necked turtle (*Chelodina oblonga*), and various native birds (Dooley *et al.*, 2006). A total of 19 fauna species of conservation concern have been recorded within the site (DBCA, 2025c).

This context offers a biodiverse but potentially challenging environment for enhancing connectivity, driven by highly fragmented native vegetation outside protected areas and significant physical barriers. An opportunity exists to understand how to plan and prioritise rehabilitation of habitat connectivity in an established urban environment.

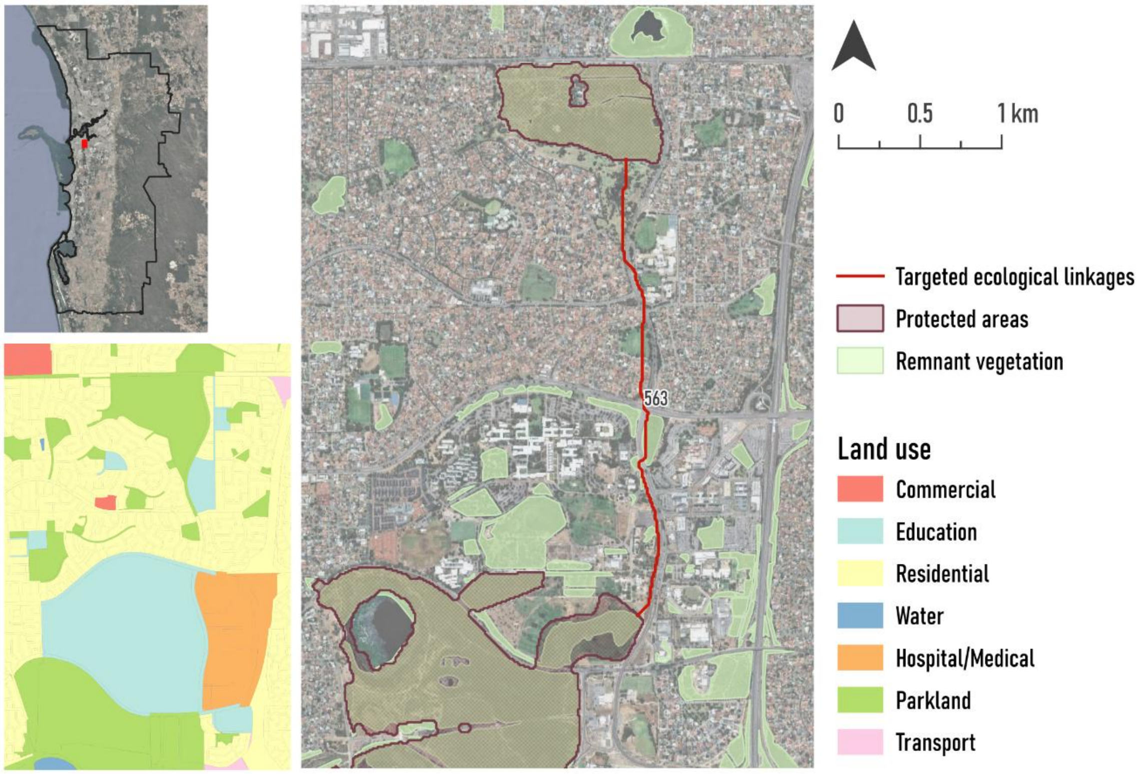


Figure 21. Overview of the existing urban site, showing targeted ecological linkages, protected areas, and patches of remnant vegetation, with the linkage ID shown along the linkage path (right panel). The upper left panel shows the site context within the broader region, while the lower left panel shows the land uses within the site. Legends are provided to the right of the panels. Base maps: Imagery © 2026 Google, Airbus, CNES / Airbus, Maxar Technologies, Vexcel Imaging US.

Emerging urban

The emerging urban site spans approximately 28km^2, with its boundary just over 3km southeast of the existing urban site. However, it varies greatly in development context, sitting at the urban fringe where housing development is actively occurring (Figure 22). Two high-priority ecological linkages were selected for this case study, joining a protected area to the east of the site to an intermediate reserve (linkage 654), and from this reserve to a protected area in the northwest (linkage 624). Multiple other protected areas are present, including four in the southwest and one in the northeast, though the linkages between these areas were slightly lower priority. Both linkages scored highly across all categories, though linkage 624 scored slightly lower in conservation values than linkage 654 (72.5 vs 83.4), indicating the southeast protected area was a high priority for conservation (Table 9).

The site contains a mixture of suburban residential development, peri-urban bush lots and parkland, with aerial imagery from 2025 showing urban development was actively occurring (Landgate, 2025) (Figure 22). A 4-lane divided highway traverses the south of the site in an east-west direction, while the two ecological linkages trace a large powerline corridor through the suburban environment that consists mostly of degraded vegetation and bare soil. Peri-urban lots in the west of the site support some remnant vegetation, though patches vary in condition. Conversely, the suburban area retains small, isolated patches of remnant vegetation in parks linked by a system of vegetated open swales. This part of the site is relatively low-lying, so many of these remnant urban parks support wetland habitats (DBCA, 2024), though the original hydrology is likely to be significantly altered. The vegetation within the site is mostly characterised by banksia and eucalypt woodlands, with low-lying areas and wetlands supporting melaleuca woodland communities (DBCA, 2018b). All but one of the protected areas (west of the site above the suburban development), and even some of the suburban parks, contain conservation category wetlands (DBCA, 2024). Together they support a wide range of biodiversity, with 16 conservation significant fauna species recorded in the site including quenda (*Isoodon fusciventer*), western brush wallaby (*Notamacropus irma*), and rakali (*Hydromys chrysogaster*) (DBCA, 2025c).

The emerging urban site is distinctive because urban development here is relatively recent and ongoing, providing a landscape where patterns of fragmentation are still forming. Small patches of native vegetation have been retained to varying degrees due to the peri-urban residential lots and more contemporary urban design practices. These small remnants can be crucial for functional connectivity and are likely to alter rehabilitation priorities at this evolving site.

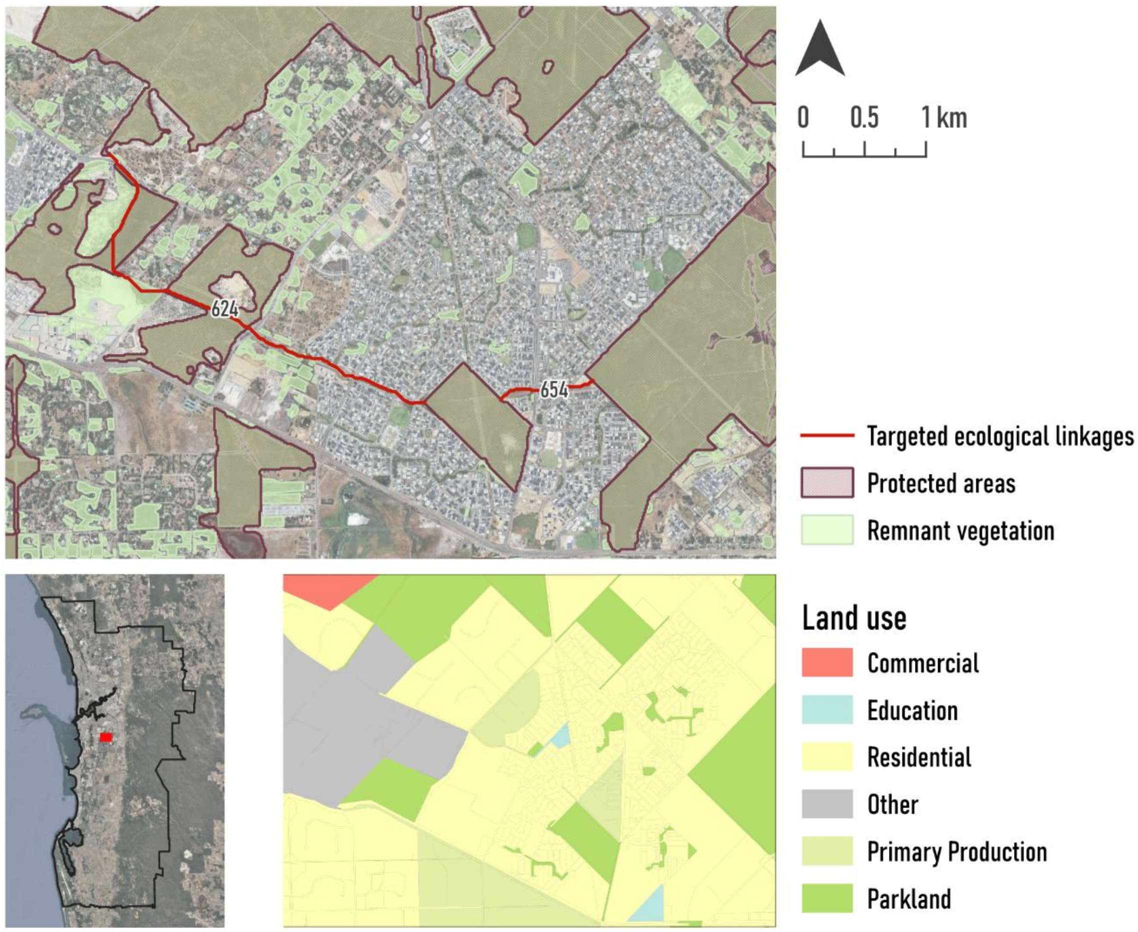


Figure 22. Overview of the emerging urban site, showing targeted ecological linkages, protected areas, and patches of remnant vegetation, with the linkage ID shown along the linkage path (upper panel). The lower left panel shows the site context within the broader region, while the lower right panel shows the land uses within the site. Legends are provided to the right of the panels. Base maps: Imagery © 2026 Google, Airbus, CNES / Airbus, Maxar Technologies, Vexcel Imaging US.

Rural

The rural case study is located approximately 85km south of Perth city centre, with the matrix primarily consisting of farmland (Figure 23). The site covers approximately 22km^2 and includes three distinct ecological linkages (1194, 1197, 1210), though all essentially connect two large patches of native vegetation that exist to the west and east of the site. The western patch is encompassed by single protected area, while the eastern patch composed of three distinct but ecologically connected protected areas. A smaller protected area exists in the centre of the site, acting as an intermediate patch joined to linkages 1194 and 1197.

Each of the three linkages scored highly across all categories. However, they scored noticeably lower for connectivity, and higher for route attributes, than linkages at the other two sites, representing a slightly more challenging traverse for wildlife but a potentially more forgiving rehabilitation context (Table 9). The two northern linkages follow roadside vegetation, while the southern linkage traverses agricultural properties following patches of remnant trees. Those linkages crossing the freeway utilise culverts.

The site is homogenous, with all land uses outside of parkland designated for agriculture (Figure 23). A major barrier is an unwalled freeway that traverses north-south to the west of the central protected area. Small patches of remnant vegetation are scattered across the site between the larger remnants, though much of this is highly degraded and represents a remnant eucalypt overstory over pasture weeds. Compared to the other sites, there are large distances between adequate stepping stone habitats. All protected areas contain conservation category wetlands (DBCA, 2024), with the western patch also associated with a Ramsar wetland of international importance (DBCA, 2017b). The site contains a range of habitats, including variations of eucalypt forests and melaleuca woodlands in low-lying areas (DBCA, 2018b), with 18 recorded fauna species of conservation significance (DBCA, 2025c). Many of these species are migratory birds, however, the site has not been as intensively surveyed as the other two case studies according to publicly available biodiversity data (DBCA, 2025a). The site supports a range of native amphibian and reptile species, as well as quenda (*Isoodon fusciventer*) (DBCA, 2025a, 2025c) .

The rural site offers a contrasting connectivity challenge, with large, high-value remnants separated by a relatively wide and homogenous stretch of agricultural matrix, unsupported by stepping stones. While potentially more challenging from a connectivity perspective, the site offers a rehabilitation advantage with limited hard surfaces and less intensive development.

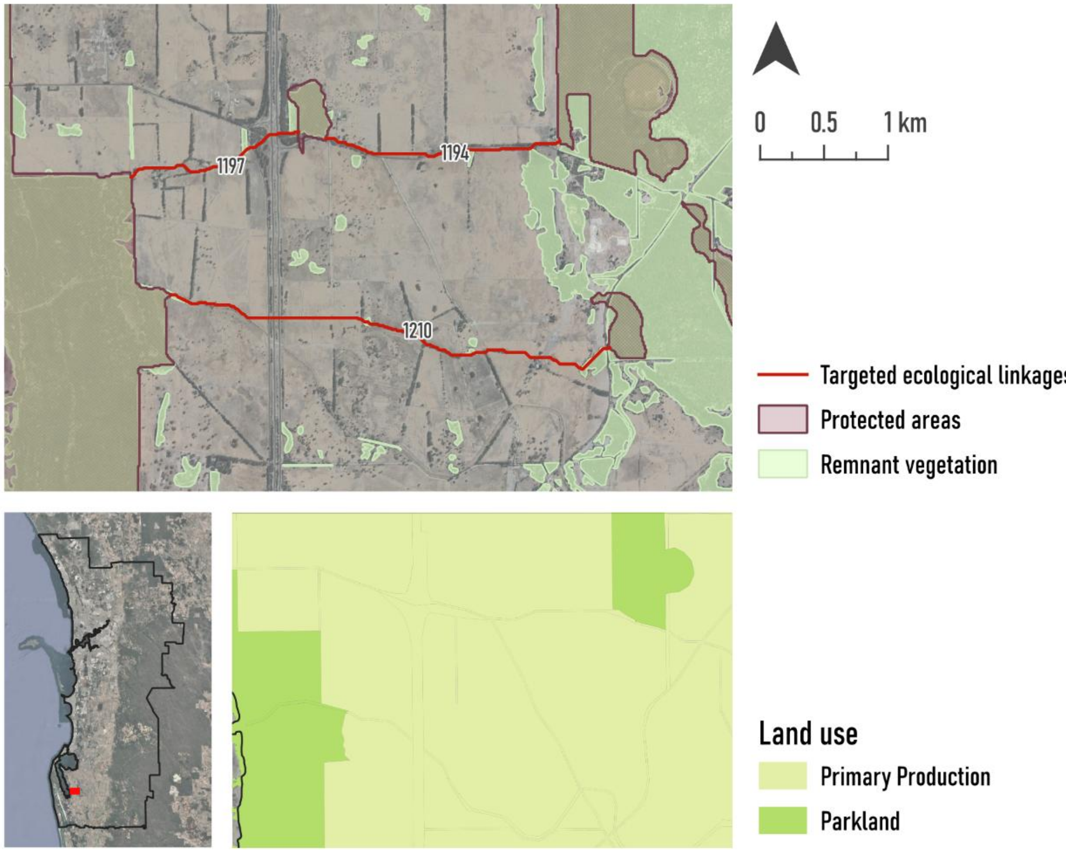


Figure 23. Overview of the rural site, showing targeted ecological linkages, protected areas, and patches of remnant vegetation, with the linkage ID shown along the linkage path (upper panel). The lower left panel shows the site context within the broader region, while the lower right panel shows the land uses within the site. Legends are provided to the right of the panels. Base maps: Imagery © 2026 Google, Airbus, CNES / Airbus, Maxar Technologies.

4.2 Methods

4.2.1 Overview and wildlife guild development

To identify rehabilitation priorities at a local scale, baseline patterns in connectivity were evaluated for four wildlife guilds within the three study areas. This understanding of existing functional connectivity and dispersal limitations for species of differing capability is necessary to interpret where and how rehabilitation may best support improved organism movement (Saura *et al.*, 2014; LaPoint *et al.*, 2015; Lynch, 2018).

Four wildlife guilds were identified to represent functional connectivity of four sets of species with distinct habitat requirements and dispersal abilities (Table 10). Groupings were chosen to represent a range of distinct habitat niches and movement abilities for common fauna in the Perth region. The *tree birds* guild represents larger birds that depend on native tree species, with moderate to large dispersal distances, including two iconic conservation-significant black cockatoo species. The *woodland birds* guild represents smaller bushland birds, such as wrens, which are more dispersal limited and rely on dense shrub layers for habitat. The *terrestrial fauna* guild represents small ground-dwelling vertebrates, while the *wetland fauna* guild represents fauna with a distinct requirement for wetland habitats, which are likely sparser in the landscape than terrestrial habitats. The guild approach recognised variability in habitat needs and movement abilities, while enabling potential rehabilitation interventions to benefit a range of species. A review of the literature was used to define representative species from each guild and to generalise their habitat and movement ecologies – a summary of which is provided in Appendix B.

Table 10. Wildlife guilds focused on for the connectivity analyses, including a description, selected dispersal threshold and list of five representative species. Species marked with an (*) are listed as priority or threatened under state or federal legislation. A review of the habitat and movement ecology of each guild used to inform descriptions and dispersal thresholds are provided in Appendix B.

Guild	Description	Dispersal threshold	Representative species
 Tree birds	Moderately sized birds that rely on mature trees – particularly eucalypts or Proteaceous species – for nesting, foraging or roosting.	6km	• Carnaby's Black Cockatoo (*Calyptorhynchus latirostris*)* • Red-tailed Black Cockatoo (*Calyptorhynchus banksii naso*)* • Grey Butcherbird (*Cracticus torquatus*) • Tawny Frogmouth (*Podargus strigoides*) • Tree Martin (*Hirundo nigricans*)
 Woodland birds	Small bushland birds with limited dispersal ability that require dense shrub layers for nesting and shelter.	400m in open grassland	• Inland thornbill (*Acanthiza apicalis*) • Splendid Fairy-wren (*Malurus splendens*) • Western Spinebill (*Acanthorhynchus superciliosus*) • Western Wattlebird (*Anthochaera lunulata*) • White-browed Scrubwren (*Sericornis frontalis*)
 Terrestrial fauna	Small ground-dwelling mammals and reptiles that require intact understorey or litter for shelter and foraging.	400m in urban areas	• Bobtail (*Tiliqua rugosa*) • Perth Slider, Lined Skink (*Lerista lineata*)* • Quenda (*Isoodon fusciventer*)* • Soft spiny-tailed gecko (*Strophurus spinigerus*) • Western brush wallaby (*Notamacropus irma*)*
 Wetland fauna	A range of vertebrate fauna that depend on wetlands for some or all stages of their life cycle.	500m in vegetated areas	• Australasian bittern (*Botaurus poiciloptilus*)* • Clicking froglet (*Crinia glauerti*) • Moaning frog (*Heleioporus eyrei*) • Snake-necked turtle (*Chelodina oblonga*) • Water rat, Rakali (*Hydromys chrysogaster*)*

* Species of conservation concern

Baseline functional connectivity was assessed for each guild within the three study areas using Circuitscape 5 (McRae *et al.*, 2008; Anantharaman *et al.*, 2020). Three types of input layers were constructed for the Circuitscape analyses – a focal layer, resistance layer, and ground layer. The *focal layer* was a vector layer representing patches of habitat or stepping stones (focal patches), which were later converted to raster sources or destinations as required by Circuitscape. Current flowed from sources over a *resistance layer*, a heterogenous raster surface describing the relative movement difficulty over different landscape features. The *ground layer* was a raster that controlled current leakage out of the system at each step in the landscape, which served two functions. Firstly, it set all non-source focal patches as movement destinations, towards which current flows to exit the system entirely. Secondly, it applied a small degree of current leakage to the rest of the landscape, occurring at each step to represent movement attrition (from mortality or settling) of organisms dispersing between habitats. This cumulatively caused exponential decay of movement potential with distance and was parameterised for each guild to represent their dispersal thresholds. These input layers were generated separately for each guild and site to reflect differences in landscape attributes, movement abilities and habitat requirements.

The connectivity modelling occurred in four stages (Figure 24). Firstly, preliminary input layers were constructed using rule-based spatial processing. These layers were then refined through systematic desktop and targeted field assessments to overcome data limitations and improve mapping accuracy. Circuitscape analyses were then conducted to produce current density maps which represent movement probability across the landscape. Finally, a framework was developed to identify rehabilitation priorities from the outputs. This framework outlined key management objectives to improve connectivity, then provided guidance to identify spatial priorities for each objective. To further support the framework, two additional outputs were derived from the current density maps: details on the emigration success from each focal patch, and candidate areas for rehabilitation. The framework provided a transparent way to identify rehabilitation priorities, drawing on each of the outputs for ecological context.

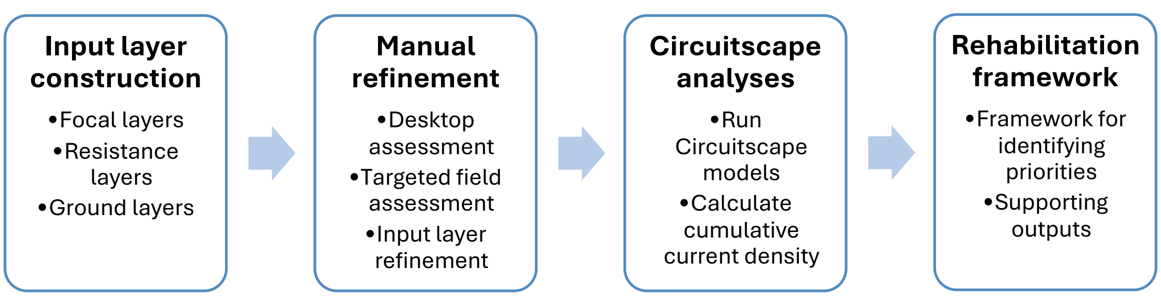


Figure 24. Overview of the local connectivity assessment methodology used in this study.

4.2.2 Input layer construction

Inputs to Circuitscape analyses need to reflect the characteristics of the focal organisms. For each guild, therefore, a review of the habitat and movement literature of relevant species (Appendix B) was used to define inputs for the connectivity analyses. This included defining the ecological criteria for a focal patch (patches of habitat or stepping stones), the relative resistance of various landscape types, and the 'resistance to ground' value which modulates the ground layer to model dispersal limitations of each guild. Preliminary versions of each layer were constructed before manual desktop and field refinements.

Input data

A range of spatial data were acquired and processed to build the input layers (Table 11). All datasets used are listed together here, as some of the data were used in the development of multiple input layers. Detailed information on the processing of initial data layers is provided in Appendix C.

For the focal and ground layers, a range of vector layers were constructed to identify focal patches (habitat or stepping stones). For the resistance layers, data represented a range of landscape features affecting wildlife movement, including vegetation structure, land cover, land use, and major barriers (Table 11). A primary dataset used for all input layers was a detailed canopy height model (CHM) built from 2020 Urban Monitor vegetation height data (Caccetta *et al.*, 2012, 2016), which helped to describe vegetation structure across the case study sites. To simplify processing and match the analysis resolution of the connectivity modelling, the CHM was upscaled from 0.2m to a 10m horizontal resolution, and used to represent simple presence/absence of vegetation at 5 different height strata (Table 11). A limitation of the CHM data was in its ability to identify understorey vegetation, given it only measured the top layer of vegetation. Some of this was able to be overcome in the process of upscaling the data, by virtue of detecting the uppermost layer of understorey between gaps in tree canopy, though understorey presence was often unrecognised where tree cover was dense. In some instances, the CHM also did not identify any vegetation where recent aerial imagery confirmed its presence (Landgate, 2025). To fill these gaps, which would appear in the resistance layers, Sentinel-2 NDVI imagery was used to identify perennial or annual vegetation, albeit without height information. Like the CHM layers, all input resistance layer data representing the landscape features listed in Table 11 were converted to presence/absence rasters at 10m resolution. This resolution provided the detail necessary to capture relatively fine-scale landscape structure while reducing computational complexity and aligning with available satellite imagery.

Table 11. Landscape features and datasets used for the construction of focal, resistance and ground layers of all guilds. Detailed processing information for each landscape feature is provided in Appendix A.

Landscape feature	Description	Datasets
Focal patches (vector data used in focal and ground layers)		
Native vegetation patches	Remnant native vegetation polygons that have undergone digital smoothing to represent contiguous habitat patches	Native vegetation extent (DPIRD, 2023) *Habitat barriers* feature
Black cockatoo nesting and roosting sites	Patches of tall trees surrounding known nesting or roosting locations of Carnaby's or forest red-tailed black cockatoos (*C. latirostris*, *C. banksii naso*)	Threatened species and communities database (DBCA, 2025c) Urban Monitor (Caccetta *et al.*, 2012, 2016) vegetation height data for 2020.
Conservation category wetlands	Wetlands designated in the 'conservation' management category by DBCA under state policy (DBCA, 2025d)	Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024)
Protected areas (NatureLink)	Protected areas as defined by O'Donnell (2020)	NatureLink Perth protected areas data (O'Donnell, 2020)
Habitat barriers	Roads and railway lines used to separate otherwise contiguous focal patches	Road Network (Main Roads, 2025) OpenStreetMap rail data: July 2025 (OpenStreetMap Contributors, 2025)
Vegetation strata (raster data used for all input layers)		
Tall trees	Presence of tall trees (>8m)	Urban Monitor (Caccetta *et al.*, 2012, 2016) vegetation height data for 2020.
Trees	Presence of trees (3-8m)	
Shrubs	Presence of shrubs (1.5-3m)	
Subshrubs	Presence of subshrubs (0.5-1.5m)	<u>Perennial groundcover:</u> Sentinel 2B satellite imagery: dates 6/08/2024 and 1/03/2025 (ESA, 2025) Random forest-classified lawn data sourced from Peterson and Andrew (2025)
Groundcover	Presence of perennial groundcover (0-0.5m) (this was filtered to distinguish perennial vegetation from annual weeds and lawn, which were otherwise recognised in the groundcover stratum)	

Land cover (raster data used in resistance layers)		
Remnant vegetation	Native remnant vegetation or native regrowth – this includes native vegetation that is uncleared or partially cleared after European settlement, and excludes planted vegetation unless planted for conservation purposes under legal requirements (DBCA, 1986 s. 51A)	Native vegetation extent (DPIRD, 2023)
'Other' perennial vegetation	Presence of perennial vegetation, including deciduous trees, without height information	Sentinel 2B satellite imagery: dates 6/08/2024 and 1/03/2025 (ESA, 2025) Random forest-classified lawn data sourced from Peterson and Andrew (2025) Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024)
Annual vegetation	Presence of annual vegetation (grasses, herbs), including summer-green wetland vegetation, without height information	
Bare surfaces	Non-vegetated areas (bare soil, asphalt, roofs)	
Open water*	Exposed surface water from the *DEA Water Observations (Sentinel-2 NRT)* dataset	Water Observations from Space: date 21/12/2024 (Geoscience Australia, 2024)
Wetland water	Surface water within wetlands that would have otherwise been classified as 'open water'. This layer enabled the identification of water associated with a wetland system and supports movement of wetland fauna.	Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024) Water Observations from Space: date 21/12/2024 (Geoscience Australia, 2024)
Land use (raster data used in resistance layers)		
Additional protected areas	Land designated with a level of environmental protection This ensured to include a range of areas such as smaller Bush Forever sites, Regional Parks, and cleared vegetation, which fell under protected land uses but were excluded from the original protected area dataset (O'Donnell, 2020)	Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024) Native vegetation extent (DPIRD, 2023) Bush Forever (DPLH, 2019) Regional Parks (DBCA, 2023) Ramsar Sites (DBCA, 2017b) Legislated Lands and Waters (DBCA, 2025b)
Drainage channels	Open drainage channels	Drainage Open Channel (Water Corporation, 2025)
Airports	Land areas and infrastructure used for aviation	OpenStreetMap aeroway data: July 2025 (OpenStreetMap Contributors, 2025)
Parks and forestry	Public green space and managed forestry areas	Urban forest mesh blocks (DPLH, 2025)
Rural	Primary production land and agricultural areas	
Street block	Residential or commercial areas	

Campus	Institutional grounds for education, hospitals and some recreation	
Industrial	Land zoned for industrial operations	
Other	Some transport and utility areas, including road corridors	
Barriers (raster data used in resistance layers)		
Transmission overhead powerlines	Large, high-voltage transmission powerlines supported by towers	Transmission Overhead Lines (Western Power, 2025)
Rail corridors	Land reserved for rail and train operations	OpenStreetMap rail data: July 2025 (OpenStreetMap Contributors, 2025)
State roads	Major state government-owned roads designed for large volumes of traffic	Road Network (Main Roads, 2025)
Local roads	Local government-owned roads designed for local traffic at low to moderate volumes	
Culverts	Under-road tunnels or channels designed for water drainage	Culverts (Main Roads, 2024) Road Network (Main Roads, 2025)
Buildings	Permanent, enclosed human-made structures	Buildings of WA (DPIRD, 2022)

* 'Open water' is listed under the barriers category for terrestrial fauna

Focal layers

Focal layers consisted of a set of distinct focal patches to be used as sources and destinations of current. These patches represented both larger habitats and smaller stepping stones, and were identified for each guild using ecological criteria relevant to their habitat requirements (Table 12). These criteria allowed for stepping stones to be considered as focal patches, acknowledging their role in supporting organism dispersal (Saura *et al.*, 2014) and therefore enabling movement between areas where current could have otherwise dissipated due to movement attrition.

Preliminary versions of the focal layers were constructed first, using rule-based criteria (Table 12) that were applied to available spatial data (Table 11). These preliminary layers intended to serve as approximations of potential focal patches, that could be validated through the desktop and field refinement process. The primary input, used for three of the four guilds, was a pre-processed layer called *native vegetation patches*, which represented patches of contiguous remnant vegetation with potential to provide suitable habitat (Table 11). For wetland fauna, conservation category wetlands, or any wetlands within protected areas, were used instead. These patches were then filtered using summaries calculated from the CHM to identify whether they contained sufficient vegetation cover at the strata suitable for the relevant guilds. For example, tree birds required the presence of trees or tall trees, while terrestrial fauna required subshrubs or groundcover (Table 12). For tree birds, focal patches also included known black cockatoo nesting or roosting sites. The resulting polygons all represented potential patches of contiguous habitat for each guild, separated by barriers or large gaps. Each focal patch was assigned a unique ID number.

Table 12. Ecological criteria for inclusion as a focal patch under each guild; including quantitative rules for preliminary focal patch selection. Vegetation condition was defined using the Keighery vegetation condition scale (Keighery, 1994). Italicised text refers to input data features from Table 11.

Guild	Focal patch criteria	Preliminary rule-based criteria
Tree birds	Native vegetation, planted or remnant, providing structure and diversity similar to that of good or better condition, with an intact overstorey layer (>3m) including eucalypt or banksia species. Include patches of this vegetation in degraded condition if over 1ha of native overstorey is intact.	*Native vegetation patches* with >20% overstorey cover (*trees* or *tall trees*). Split by *habitat barriers*.
	Known black cockatoo nesting or roosting sites.	Any *black cockatoo nesting and roosting sites*.
Woodland birds	Native vegetation, planted or remnant, providing structure and diversity similar to that of good or better condition, with an intact midstorey layer (0.5m-3m).	*Native vegetation patches* with >10% midstorey cover (*shrubs* or *subshrubs*). Split by *habitat barriers*.
Terrestrial fauna	Native vegetation, planted or remnant, providing structure and diversity similar to that of good or better condition, with an intact understorey layer (0m-1.5m).	*Native vegetation patches* with >10% understorey cover (*subshrubs* or *perennial groundcover*). Split by *habitat barriers*.
Wetland fauna	Wetlands with native fringing vegetation, planted or remnant, providing structure and diversity similar to that of good or better condition, with an intact understorey layer (0m-1.5m).	*Conservation category wetlands*, or any wetlands within *protected areas*, with >10% understorey cover (*subshrubs* or *perennial groundcover*). Split by *habitat barriers*.

Resistance layers

Resistance layers described relative movement difficulty for a given guild at a given cell. They were constructed by combining a range of presence/absence layers representing various landscape features.

Following O'Donnell (2020), landscape features were divided into three categories: land cover, land use, and barriers (Table 13). Land cover represented the physical features on-ground that assist or hinder movement. This included features such as vegetation, bare surfaces, and open water. Vegetation structure was defined in detail based on the vegetation strata layers derived from the CHM (Table 11). For example, multi-storey vegetation could be identified where a single 10m cell showed presence of multiple vegetation strata such as groundcover, shrubs, and trees. Cells containing this vegetation were further defined as remnant or non-remnant based on whether or not they intersected with state-wide remnant vegetation mapping (DPIRD, 2023). The land use category signified the designated human function of the land, providing an approximation of the intensity of human activity. This included land uses such as parks and forestry, street blocks, or campuses. The barriers category included major landscape features that hinder movement. These generally consisted of human infrastructure such as roads or buildings, or natural elements such as open water.

The resistance values assigned to each landscape feature were unique to each guild, based on relevant habitat and movement ecology literature (Appendix B). Studies developing state-space models or step-selection functions from quantitative movement data are ideal for the development of the resistance layers, though literature for representative species included a lack of such studies. While telemetry-based movement data were available for limited species (Robinson *et al.*, 2018; Rycken *et al.*, 2022), habitat selection and distribution modelling studies, global databases and local guide books with habitat and behavioural observations, and mortality-related studies were utilised in lieu of more robust studies to estimate resistance values. Observing recommendations by Beier *et al.* (2011), landscape features were generally assigned a resistance value between 1-100. Barriers were assigned a resistance of 100 or more, with a maximum of 500 for highly impermeable landscape features. For tree birds, a lower range of resistance values was utilised for landscape features excluding airports, due to their greater ease of movement over most landscape features compared to the other guilds.

To construct the resistance layers, three composite rasters – land cover, land use, and barriers – were created by overlaying the relevant landscape features parameterised by their guild-specific resistance value. This was done in order from bottom to top as per Table 13. Each layer was 'burnt in' to overwrite the layer beneath, so that the resistance value on top of the list took highest priority. These three rasters were then combined using a weighted average of land cover (weight = 2) and land use (weight = 1), masked by the barrier layer. This formed a single resistance layer which accounted for both land cover and land-use effects, while placing more importance on the physical landscape than its particular use. The final barriers overlay ensured significant barriers were always included in the final resistance layer and that the resistance value due to barriers was not modulated by land cover or land use.

Table 13. Resistance values assigned to landscape features for each guild, organised by land cover, land use and barrier categories. Resistances not assigned to a landscape feature for a particular guild are marked with a dash (-). The order of landscape features within each category correspond to the order in which they were overlaid in the intermediate layers (top = overlaid last). Italicised text refers to input data features from Table 11.

Landscape feature		**Resistance**			
Feature	**Description**	**Tree Birds**	**Woodland birds**	**Terrestrial fauna**	**Wetland fauna**
Land cover					
Remnant: tall trees	*Remnant vegetation* with *tall trees* that are likely to provide roosting, nesting or foraging resources	1	-	-	-
Remnant: trees	*Remnant vegetation* with *trees* that are likely to provide roosting or foraging resources	5	-	-	-
Remnant: midstorey or understorey only	Midstorey or understorey vegetation (*shrubs*, *subshrubs* or *groundcover*) within *remnant vegetation*, where canopy (*trees*, *tall trees*) is absent	15	-	-	-
Non-remnant: tall trees	*Tall trees* outside of *remnant vegetation* that may provide roosting or foraging resources	10	-	-	-
Remnant: multi-layered with midstorey	*Remnant vegetation* including both a midstorey layer (*shrubs* or *subshrubs*) and other vegetation layers (*groundcover*, *trees*, *tall trees*)	-	1	-	-
Remnant: midstorey shrubs	Midstorey (*shrubs* or *subshrubs*) within *remnant vegetation*, where canopy (*trees*, *tall trees*) is absent	-	5	-	-
Remnant: canopy only	Canopy (*trees*, *tall trees*) within *remnant vegetation*, where midstorey and understorey (*shrubs*, *subshrubs*, *groundcover*) are absent	-	30	-	-
Non-remnant: midstorey shrubs	Midstorey (*shrubs*, *subshrubs*) vegetation outside of *remnant vegetation*	-	15	-	-
Non-remnant: canopy only	Canopy (*trees*, *tall trees*) outside of *remnant vegetation*, where midstorey and understorey (*shrubs*, *subshrubs*, *groundcover*) are absent	-	50	-	-
Remnant: multi-layered with understorey	*Remnant vegetation* including both an understorey layer (*subshrubs* or *groundcover*) and taller vegetation (*shrubs*, *trees*, *tall trees*)	-	-	1	1
Remnant: understorey only	Understorey (*subshrubs* or *groundcover*) within *remnant vegetation*, where taller vegetation (*shrubs*, *trees*, *tall trees*) is absent	-	-	5	5
Remnant: tall shrubs or canopy only	Tall shrubs or canopy (*shrubs*, *trees*, *tall trees*) within *remnant vegetation*, where understorey (*shrubs* or *subshrubs*) is absent	-	-	40	30
Non-remnant: understorey	Any understorey vegetation (*shrubs* or *subshrubs*) outside of *remnant vegetation*	-	-	15	15
Non-remnant: tall shrubs or canopy only	Tall shrubs or canopy (*shrubs*, *trees*, *tall trees*) outside of *remnant vegetation*, where understorey (*shrubs* or *subshrubs*) is absent	-	-	60	60
Wetland water	Open water within perennial or seasonal wetlands with potential to provide habitat resources	-	-	-	5
Open water	Permanent surface water	30	90	-	20

Landscape feature		Resistance			
Feature	**Description**	**Tree Birds**	**Woodland birds**	**Terrestrial fauna**	**Wetland fauna**
Remnant: other perennial vegetation	*'Other' perennial vegetation* (not CHM-derived) within *remnant vegetation* areas	15	30	40	40
Remnant: annual vegetation	*Annual vegetation* (grasses, weeds) within *remnant vegetation* areas	30	70	60	60
Non-remnant: other perennial vegetation	Other *perennial vegetation* of any height outside of *remnant vegetation* areas	25	50	60	60
Non-remnant: annual vegetation	*Annual vegetation* (grasses, crops, lawn) outside of *remnant vegetation* areas	30	80	80	80
Bare	Non-vegetated areas (asphalt, paving, bare soil, and roofs)	30	90	90	90
Land use					
Airport airspace	Areas within 500m of *airports*	100	-	-	-
Protected areas	Land designated with a level of environmental protection	1	1	1	1
Utilities - drainage channels	Open drainage channels	20	20	20	5
Parks and forestry	Public green space and managed forestry areas	10	10	10	10
Rural	Primary production land and agricultural areas	30	40	40	40
Street block - residential or commercial	Residential or commercial areas	30	60	60	60
Campus	Institutional grounds for education, hospitals and some recreation	30	50	50	50
Industrial	Land zoned for industrial operations	40	70	80	80
Airport	Land areas and infrastructure used for aviation	-	100	100	100
Other	Some transport and utility areas, including road corridors	40	70	70	70
Barrier					
Open water	Permanent surface water	-	-	200	-
Culverts	Under-road tunnels or channels designed for water drainage	-	-	100	100
Transmission overhead powerlines	Large, high-voltage transmission powerlines supported by towers	70	-	-	-
Rail corridor	Land reserved for rail and train operations	50	150	200	200
State road	Major state government-owned roads designed for large volumes of traffic	50	150	200	200
Local road	Local government-owned roads designed for local traffic at low to moderate volumes	40	120	120	120
Building	Permanent human-made dwellings that restrict movement	-	120	200	200

Ground layers

The ground layer controls current leakage out of the system by assigning a resistance to ground (R_g) value to each raster cell. Its first role was to tie all cells that intersected with focal patches to ground ($R_g = 0$), which is typically used to define movement destinations towards which current will naturally gravitate. Its second role was to simulate finite dispersal ability by allowing a small proportion of current leakage at each step. In Circuitscape, current flowing through each cell subdivides and can move to any of its eight neighbours or 'leak' to ground (leaving the system), with the proportion of current flowing in each direction determined by relative resistances of each path (R_1-R_8 and R_g; Figure 25). The R_g value therefore influences the proportion of current leaving the system at each step, representing the chance of mortality or settling of dispersing individuals between habitats.

Constant R_g values were applied to all non-focal patch cells to cause a small proportion of current leakage at each step. Because the R_g value was constant, leakage increased where surrounding paths (R_1-R_8) had higher resistance (e.g. barriers) and decreased where they were lower resistance (e.g. vegetation), effectively reducing or increasing dispersal distances, respectively. This may represent greater mortality due to movement across high-resistance landscape features.

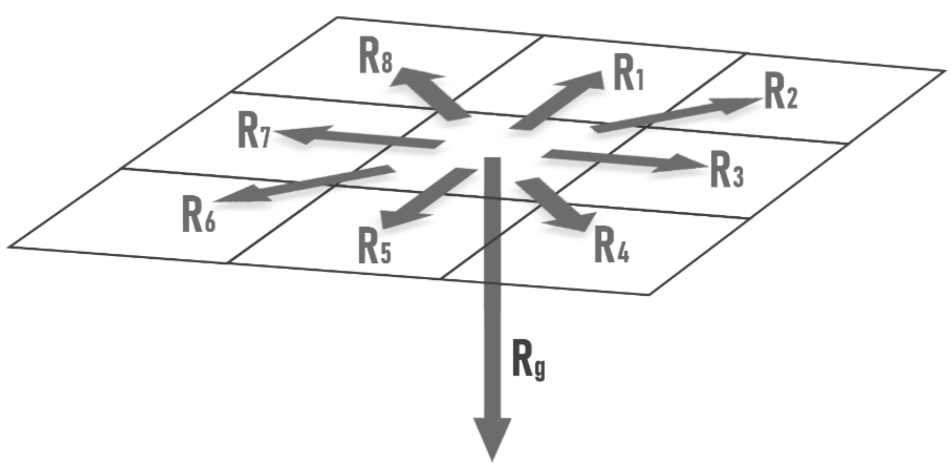


Figure 25. Diagram of raster grid cells in a Circuitscape analysis, showing the nine potential pathways for current to leave a central cell: to its neighbouring cells (1-8) or to ground (g). Arrows denote direction of potential flow, while R_x labels represent the resistance of each path.

For each guild, the constant R_g value was empirically derived through a series of Circuitscape simulations. The aim was to identify the R_g that caused 5% of the original source current to remain at a radius from the source corresponding to the guild's estimated dispersal threshold (target r_d; Table 14). Raster inputs included:

- a point-source raster with a 100A source in the centre;
- a uniform resistance raster with a constant resistance value; and
- a series of uniform ground rasters with different R_g values.

The selected dispersal threshold for each guild was based on the movement ecology of relevant species within specific landscapes, such as suburban, rural or habitat areas (Table 14). Dispersal thresholds formed the ecological basis for the target r_d, while the underlying landscape types were translated into resistance values consistent with those used in the main resistance layers (Table 13). This ensured that the R_g calibration was carried out within a realistic ecological setting. Circuitscape was then run iteratively for a variety of R_g values. For each simulation, the current remaining at increasing distances from the source was measured by summarising the total current of all cells at a given radius. This allowed the measured r_d (radius at which 5% of the original current remained) to be identified. Simulations were repeated for a range of increasing R_g values until the target r_d was surpassed. The resulting r_d values were then plotted against their corresponding R_g values, and local linear interpolation was used to estimate the R_g that produced an r_d matching each guild's actual dispersal threshold (Table 14).

To construct the ground layers, each raster was pre-loaded with the set R_g value to simulate movement attrition, while all cells within destination patches were tied directly to ground (R_g = 0), forming destination patches for current to move towards.

Table 14. Resistance to ground (R_g) values to simulate limited dispersal for four wildlife guilds in a set of Circuitscape analyses. Each R_g value was derived from a physical dispersal threshold within a corresponding landscape, with landscape resistance values approximated from Table 4. A rationale for the chosen dispersal threshold and corresponding landscape is provided.

Guild	**R_g value**	**Dispersal threshold**	**Corresponding landscape**		**Rationale**
			Type	**Resistance**	
Tree birds	160,000	5km	Any perennial vegetation (not habitat-specific) in non-remnant areas	25	This guild's dispersal threshold was based on two threatened black cockatoo species (*C. latirostris* and *C. banksii naso*), as they are of high conservation concern in the Perth region and have larger dispersal abilities than the other three guilds (DAWE, 2022). While their movement is highly variable and generally driven by food availability, travelling anywhere from 4-16.5km a day, they move slowly and follow native vegetation corridors for foraging along the way (Rycken, 2019; Rycken *et al.*, 2022). The Environmental Protection Authority WA states that foraging and water resources are needed within 6km from a roost site to maintain landscape connectivity (EPA, 2019).
Woodland birds	2625	400m	Annual vegetation in rural areas (farmland) or in urban remnants (open weedy areas)	65	Isolation thresholds for habitat patches of woodland birds in rural wheatbelt WA were consistently in the range of 300-500m apart (Watson *et al.*, 2001). Many passerines such as wrens and other bushland birds have a median dispersal distance of up to 10 home ranges from their birthplace, but cooperative breeders such as Australian wrens are highly philopatric and tend to be between 2 and 4 home range widths (Russell and Rowley, 1993). For urban superb fairy wrens (*Malurus cyaneus*) with home ranges of 1.4 ha (Parsons, 2024), this relates to a dispersal distance of approximately 170 - 530m. One study in rural WA found blue-breasted fairy wrens (*Malurus pulcherrimus*) could travel multiple kilometres between patches, however out of 302 birds over three years, only seven made these interpatch dispersals (Brooker and Brooker, 1997). It was therefore decided that 400m represented a more realistic measure of functional connectivity.
Terrestrial fauna	2500	400m	Any perennial vegetation (not habitat-specific) in non-remnant areas	60	Quenda (*Isoodon fusciventer*) had an average dispersal distance of 90m in an urban Perth site, with long range distance events of individuals spanning from 274m to a maximum of 950m (Clunies-Ross and Clark, 2011). Assuming an exponential decay in movement, where 50% of current is reached at 90m (average dispersal distance), current reaches 5% of the original at approximately 400m. Falla (2022) found urban Perth bobtail sightings were most common between 100-500m, and second most common between 500-1000m, from the edge of a reserve. This dispersal distance also sits within the dispersal range of native bush rats (*Rattus Fuscipes*) (Callandar, 2018) and a variety of reptiles in Bold Park, Perth (How, 2024).
Wetland fauna	2600	500m	Any perennial vegetation (not habitat-specific) in remnant areas	40	Southwestern snake-necked turtles (*Chelodina oblonga)* are commonly seen up to 500m from the water's edge in urban areas (Bartholomaeus, 2016), while the western swamp tortoise (*Pseudemydura umbrina)* travel up to 450m per day and have been found up to 1km outside nature reserve boundaries (Burbidge *et al.*, 2010). 500m falls within the low end of rakali (*Hydromys chrysogaster)* movements, which are up to 3km overnight through habitat (Vernes, 1998). While frog movement is not well-studied, 500m is the approximate dispersal distance of *Ranoidea raniformis* in urban Melbourne (Heard *et al.*, 2012), while other species can travel much further (Morris, 2012). Since many of the above dispersal distances were studied in more vegetated areas, a more permeable relative landscape resistance was chosen to model the resistance to ground than the other guilds.

4.2.3 Manual refinement

Manual refinement of preliminary layers was undertaken via rapid desktop and field assessments. This process took advantage of the manual input that is possible at the local scale to overcome data limitations, improve accuracy, and ensure alignment of focal patches with their ecological definitions (Table 12). The methodology was designed to be rapid and repeatable. For each site, the desktop assessments took approximately 3-5 hours each. The field assessments took approximately the same time each, excluding travel to and from the sites; however, these times varied depending on the site complexity, size, and amount of potential habitat.

The refinement process began by systematically digitising changes to the preliminary focal patches and the vegetation strata layers based on desktop assessments. The assessments were conducted using Landgate high-resolution aerial imagery (Landgate, 2025) and Google Street View imagery (Google, no date), which has increasingly established itself as an important data source in geospatial science and ecology (Olea and Mateo-Tomás, 2013; Biljecki and Ito, 2021). Uncertainty was recorded through the entire process, and areas of high uncertainty were flagged for ground-truthing in targeted field assessments. These changes were incorporated into a set of refined focal and resistance layers for the connectivity analyses. Features were digitised and attributed using consistent conventions and templates to maximise consistency and repeatability (Appendix D).

Desktop refinement

For the refinement of focal patches, key areas were identified where aerial or street view imagery suggested significant discrepancies between mapped patches and the guild's focal patch criteria (Table 12). All refinements were carried out at the level of individual polygons, which served as the analysis units for this section. Revised focal patches were digitised using consistent conventions (Appendix D) to ensure repeatability and reflect more accurate boundaries, and updated patches were then attributed with information on vegetation type, condition and presence of vegetation strata to confirm whether the feature was a focal patch.

To refine the resistance layers, key areas were identified where significant discrepancies were evident between aerial or street view imagery and the CHM-derived strata layers. A focus was placed on unrecognised understorey beneath dense canopy, perennial groundcover falsely recognised in areas of irrigated lawn, or vegetation recognised in areas cleared for recent developments after the release of the CHM data. New polygons were digitised to represent discrepancy areas, and were manually attributed with confirmed presence/absence data for each vegetation stratum, including notes where needed. Each area of newly digitised or attributed feature was assigned an uncertainty score (confident, low or high) based on the confidence in the manual digitisation and attribution. For example, the presence of understorey beneath dense canopy could only be determined through street view imagery – so where vegetation extended far beyond the roadside and made it difficult to determine the presence of understorey vegetation, a high level of uncertainty was applied. More details on this attribution are provided in Appendix D.

Targeted field assessment

Each site was visited for targeted validation of updated focal patches and discrepancy areas for vegetation strata, focusing on all features attributed as high-uncertainty and a small random sample of those set as confident and low-uncertainty. To record data in the field, QField (OPENGIS.ch, 2025) – an open-source mobile application for GIS fieldwork – was pre-loaded with these features and two additional vector layers (one point and one polygon layer) for note-taking. Each patch was traversed on foot where access was possible, and where access was restricted, visual assessments were made from public land as close as practicable, either on foot or by vehicle. Notes and geotagged photos were taken as necessary to clarify vegetation type, condition, and presence of vegetation strata. Following the field assessment, feature shapes and attributes were updated using the new information, ensuring to follow the standard procedures.

Refined input layers

The changes following the refinement process were integrated into a set of refined focal, resistance and ground layers. Focal layers were refined by filtering for features that met focal patch criteria, dissolving intersected boundaries, and assigning a unique ID to each resulting patch. Ground layers were also updated using the refined focal patches, with each being tied to ground. To refine resistance layers, the CHM vegetation strata layers were updated by overlaying them with the discrepancy area features, and the resulting refined strata layers were used in place of the originals when building the resistance layers.

4.2.4 Circuitscape analyses

A series of Circuitscape analyses were run for each guild at each site using Circuitscape 5 (Anantharaman *et al.*, 2020) via Julia v.1.11.6 (Bezanson *et al.*, 2017). In preparation for the analysis, the prepared inputs – focal, ground and resistance layers – were converted to spatially aligned rasters at 10m resolution. Focal layers were split into a series of individual source rasters, each representing a single focal patch, with each cell in the patch assigned a current value of 1A. This meant current emanating from a focal patch was scaled by patch size, ensuring that larger habitat patches contributed more to movement potential than smaller stepping stones.

Analyses were run using the advanced mode using the prepared focal, resistance, and ground layers. A total of 362 runs were conducted across all four guilds and three sites, iterating through each focal patch as the current source, with all other patches as direct ground connections. For each guild at a given site, all resulting current density maps were additively combined to create a cumulative current map. This approach emulated Circuitscape's one-to-all mode, while enabling the advanced settings required to model movement attrition and vary the current emitted between patches. This approach offered distinct benefits over traditional pairwise current flows. This included modelling simplicity (the number of individual runs for an all-to-one approach is vastly less than for a pairwise approach) and the assumption that current would naturally gravitate to nearby destination patches, rather than be forced into pairwise current flows between distant patches that are unable to be reached by short-ranging guilds.

The primary outputs of the analyses were 12 cumulative current density maps – one for each guild at each site – which represent relative movement probability through each cell in the landscape.

4.2.5 Rehabilitation framework

The framework

A framework was developed to aid in the identification of rehabilitation priorities from connectivity model outputs (Table 15). The framework begins with three management objectives which can guide rehabilitation priorities to enhance connectivity across a site. These objectives were 1) re-establishing connectivity in areas of functional disconnection, 2) enhancing low-level connectivity along key low/moderate movement pathways, and 3) supporting corridors of high movement that are degraded or hindered by barriers. To support the 'where' and the 'how' of rehabilitation, the framework provides a simple guide for identifying spatial patterns in the model outputs and relating it to specific management actions to improve connectivity. For example, bridging functional disconnection should occur where current goes to ground between patches along key movement pathways. These areas require the establishment of new high-quality stepping stones that can act as a temporary refuge for wildlife, without which organisms would be unlikely to survive the dispersal journey given dispersal limitations. In contrast, enhancing low-level connectivity should occur in diffuse flow zones or along ridges of current where predicted movement is present but limited. As organisms can already traverse between these patches to some degree, existing connectivity can be enhanced through a wider range of management actions, including increasing matrix permeability or expanding existing stepping stones.

Table 15. Framework linking rehabilitation objectives with spatial priorities and appropriate management actions. Candidate areas supporting each rehabilitation objective were identified using rule-based criteria derived from current density and resistance layers (third column).

Objective	**Spatial priorities**	**Candidate areas**	**Management action(s)**
Bridge functional disconnection	• Areas where current goes to ground between patches along potential movement pathways • Between patches that exhibit very low successful emigration	• *Near-zero movement:* Current < 0.1 A (small buffer above 0.01 A) • *Softscape:* Resistance < 75 (resistance above which the landscape is bare and likely to represent sealed surfaces or barriers) • *Open space:* Patch size at least 0.2 ha (20 contiguous cells meeting the above requirements, used to identify open space such as parks or greenways)	• Establish new stepping stones or corridors (Saura *et al.*, 2014; Lynch, 2018)
Enhance low-level connectivity	• Along key low/moderate movement pathways, including current ridges/spurs or diffuse low-moderate flow zones • Between patches that exhibit low successful emigration	• *Low/moderate movement:* Current between 0.1 A (small buffer above 0.01 A) and 75th percentile of current values • *Degraded:* Resistance > 25 (resistance of degraded native vegetation in moderate intensity land-use, for short-range guilds, e.g., no understorey in residential area) • *Not barrier:* Resistance < 100	• Improve matrix permeability (Baum *et al.*, 2004) • Increase patch area of stepping stones or strategically place new ones (Saura *et al.*, 2014; Beninde *et al.*, 2015). • Increase redundancy around highly channelled movement (Walker and Salt, 2012; Ahern, 2013).
Support degraded high-movement corridors	• Key movement pathways with degraded vegetation and moderate/high predicted movement	• *Moderate/high movement:* Current in 75th percentile of range • *Degraded:* Resistance > 20 (resistance of degraded native vegetation in low-intensity land-use typically associated with high movement, e.g., no understorey in parkland)	• Improve matrix permeability (Baum *et al.*, 2004) • Ease barriers (McRae *et al.*, 2012) • Increase redundancy around highly channelled movement (Walker and Salt, 2012; Ahern, 2013).

Secondary outputs

To support the framework in identifying spatial priorities, two additional outputs were derived from the cumulative current maps: candidate areas for rehabilitation, and the proportion of successful emigration from focal patches. Candidate areas for rehabilitation were identified by combining the current density maps with the underlying resistance layers. Using rule-based processing, candidate maps highlighted raster cells that met the required combination of predicted movement and landscape resistance for each management objective (Table 15, third column). For instance, to bridge functional disconnection, candidate areas included sites suitable to establish stepping stone habitat. These areas needed to exhibit near-zero movement, be softscape (resistance < 75), and be at least 0.2ha (20 cells), which selected for greenspace with higher rehabilitation feasibility and sufficient area to provide adequate resources and shelter for temporary settlement (Saura *et al.*, 2014; Klaus and Kiehl, 2021). Candidate areas for enhancing low-level connectivity were identified in areas of low-moderate current, but had no size requirement, and instead focused on land that was degraded but not a barrier, such as a road or a building. Candidate areas for supporting high movement focused on areas where current flow was in the 75th percentile of the total range, while including both degraded land and barriers. Where current density maps show channelled existing movement, adding redundancy to the network was provided as an additional management action. Establishing new stepping stones or increasing matrix permeability around the channelled movement can provide alternative pathways and increase movement redundancy, which can in turn enhance network resilience (Walker and Salt, 2012).

The second set of supporting outputs were maps of patch emigration success. This patch-level metric describes the proportion of total current emitted from a source patch that successfully reaches any destination patch. It highlighted patches that contributed little or no successful movement to the wider network and may therefore benefit from rehabilitation in the surrounding landscape.

$$Patch\ emigration\ success = \frac{I_s}{\sum I_d} \times 100\%$$

The total current emitted from a source patch (I_s) is equal to the number of cells in that patch, as each cell emitted 1A of current. The total current received by destination patches (I_d) was calculated using the individual current maps generated for each source patch. As current enters a destination patch just inside its boundary before going to ground, I_d was obtained by summing the current values of all cells located within destination patches.

Together, patch emigration success and candidate areas provided complementary metrics to support the primary current density maps. These supporting metrics both aid in locating rehabilitation priorities and justify recommended management actions within the interpretation framework (Table 15).

4.3 Results

4.3.1 Model outputs and rehabilitation framework

The framework produced outputs indicating movement patterns, candidate rehabilitation areas, and patch emigration success. Together, these outputs describe potential guild movement patterns and enabled identifying rehabilitation priorities for enhancing connectivity of the three short-ranging guilds (woodland birds, terrestrial fauna, and wetland fauna). Examples of these outputs are available to be explored in more detail online via an interactive platform (https://arcg.is/11bKGe1).

The panels in Figure 26 to Figure 28 summarise these outputs for the three short-ranging guilds at each site. The first column shows the current density maps, depicting relative probability of movement at each cell in the landscape, with colour fading to black at the guilds' estimated dispersal threshold. The second column highlights the rehabilitation candidate areas, identifying suitable cells for each management objective. Potential stepping stone sites to bridge functional disconnection are shown in green; areas with low movement and potential for rehabilitation in yellow; and degraded high-movement zones shown in red. The third column shows the percentage of successful emigration from each patch, with highly isolated patches (<10% successful emigration) shown in in yellow, orange, and red.

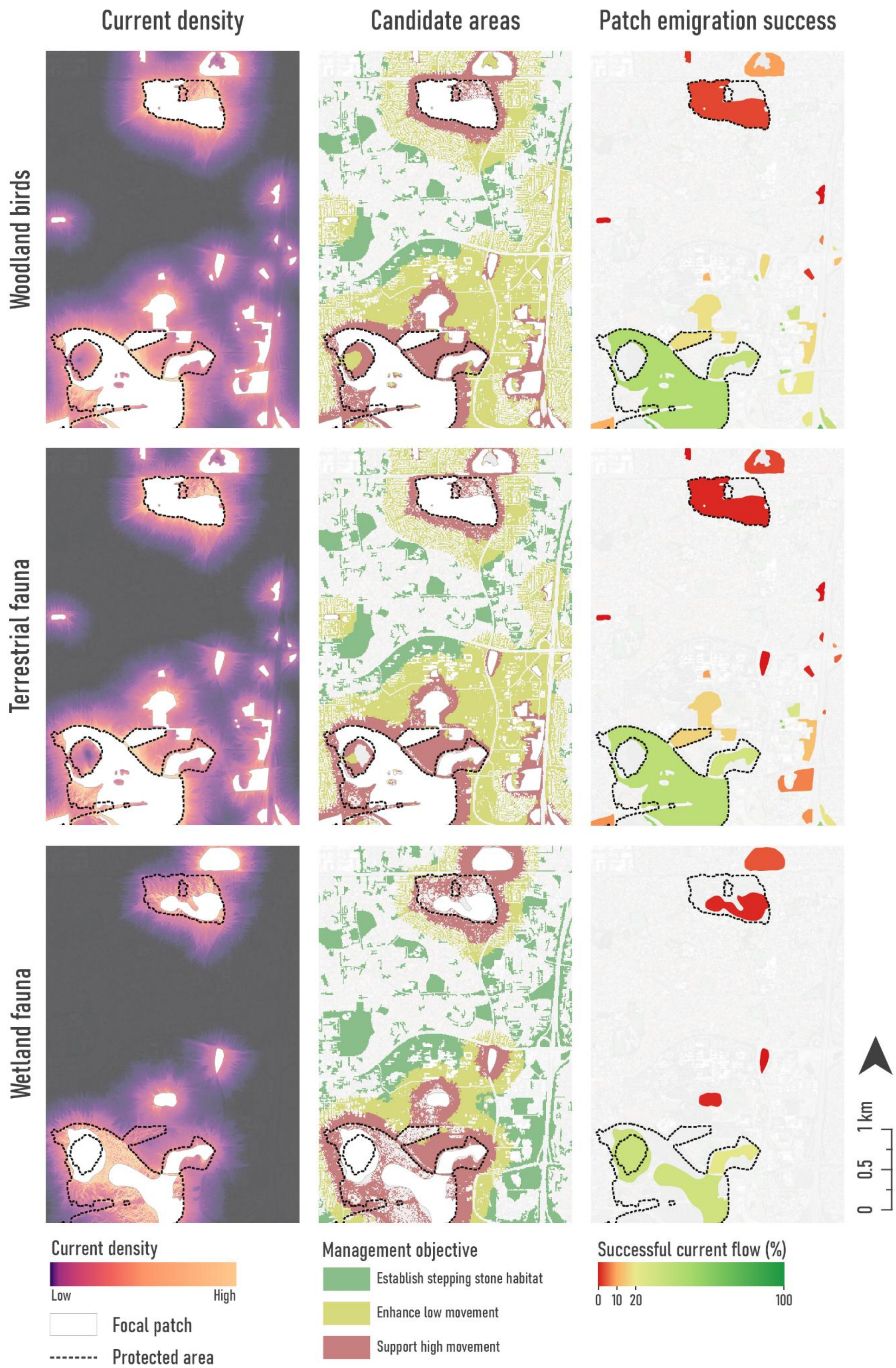


Figure 26. Outputs of the connectivity analysis at the existing urban site, enabling rehabilitation priorities to be identified for three ecological guilds. Maps show (columns) current density, rehabilitation candidate areas, and patch emigration success, presented for (rows) woodland birds, terrestrial fauna, and wetland fauna. Current density maps illustrate the distribution of movement probability. Rehabilitation candidate maps provide management objectives for all cells based on movement and underlying landscape resistance. Patch emigration success maps show the proportion of current emitted from each patch that successfully reaches another patch. Legends for each map type are provided below, with consistent colour stretches used across panels.

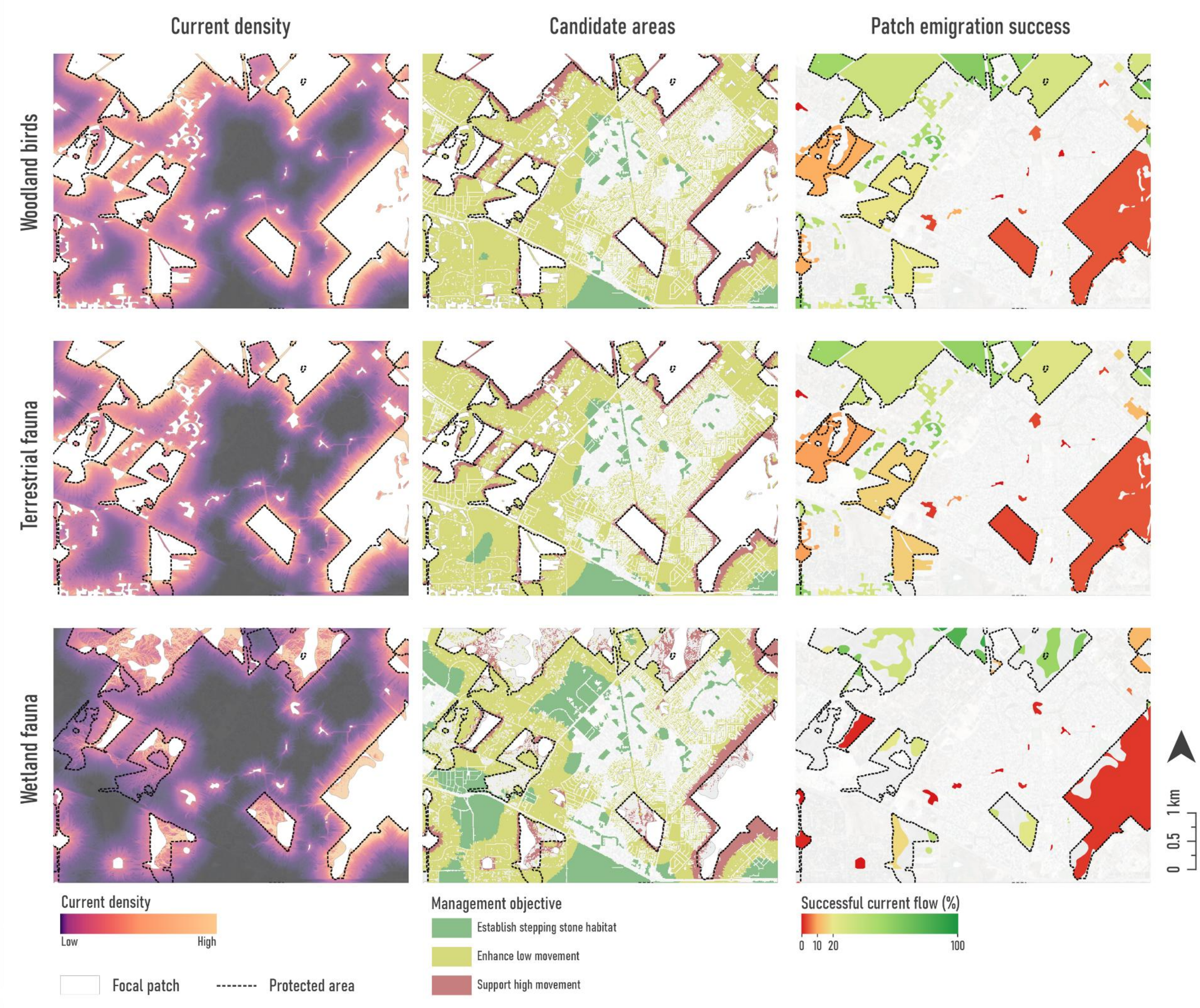


Figure 27. Outputs of the connectivity analysis at the emerging urban site, enabling rehabilitation priorities to be identified for three ecological guilds. Maps show (columns) current density, rehabilitation candidate areas, and patch emigration success, presented for (rows) woodland birds, terrestrial fauna, and wetland fauna. Current density maps illustrate the distribution of movement probability. Rehabilitation candidate maps provide management objectives for all cells based on movement and underlying landscape resistance. Patch emigration success maps show the proportion of current emitted from each patch that successfully reaches another patch. Legends for each map type are provided below, with consistent colour stretches used across panels.

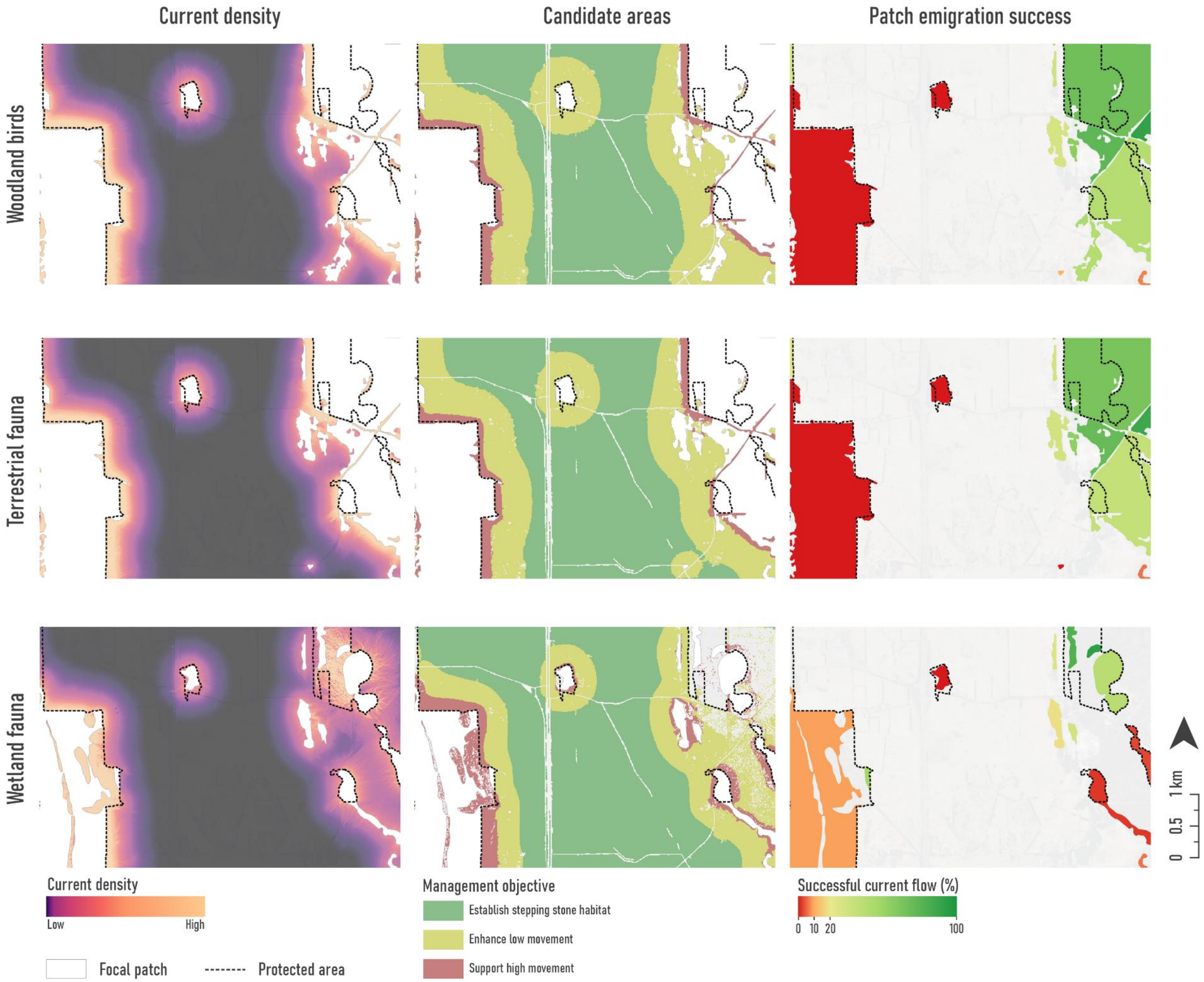


Figure 28. Outputs of the connectivity analysis at the rural site, enabling rehabilitation priorities to be identified for three ecological guilds. Maps show (columns) current density, rehabilitation candidate areas, and patch emigration success, presented for (rows) woodland birds, terrestrial fauna, and wetland fauna. Current density maps the distribution of movement probability. Rehabilitation candidate maps provide management objectives for all cells based on movement and underlying landscape resistance. Patch emigration success maps show the proportion of current emitted from each patch that successfully reaches another patch. Legends for each map type are provided below, with consistent colour stretches used across panels.

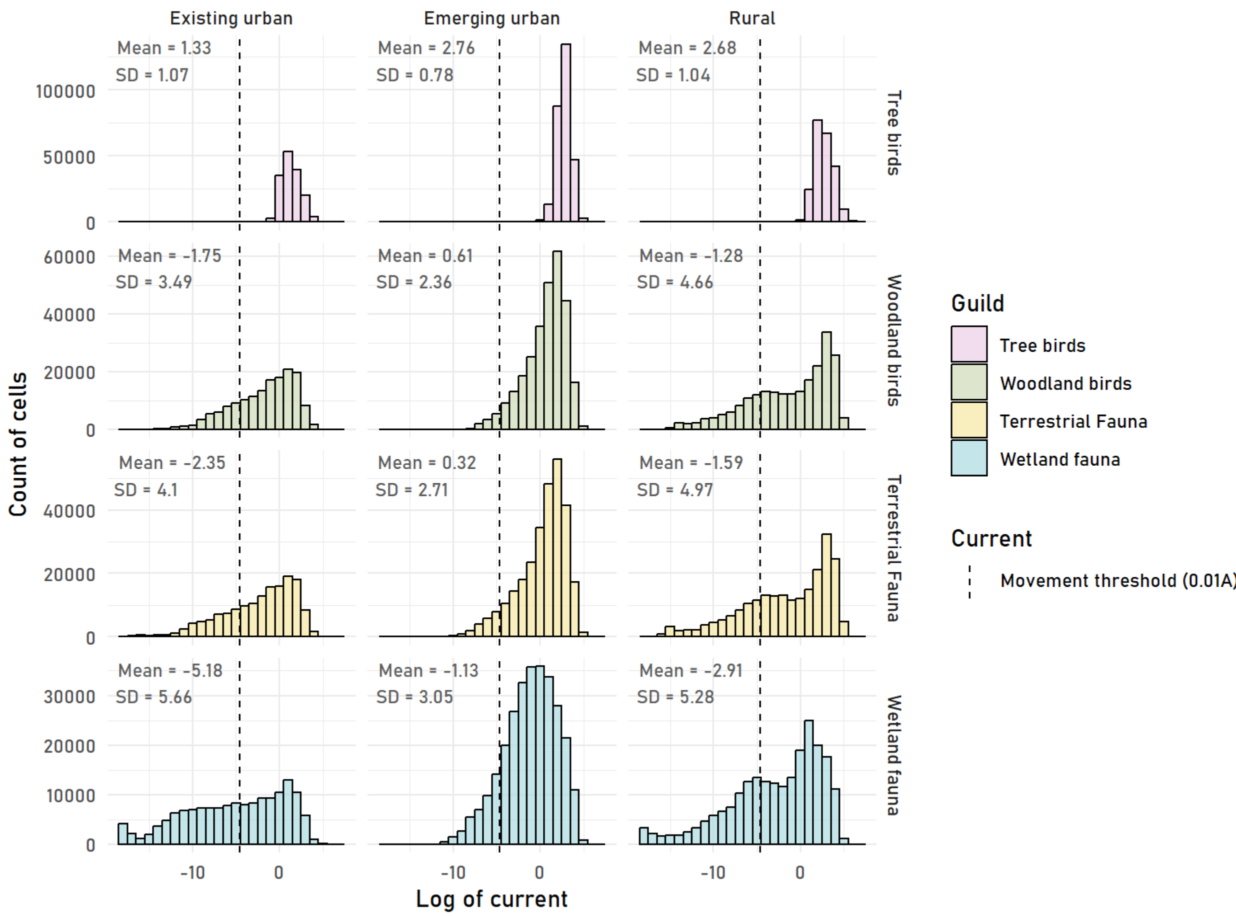


Figure 29. Distribution of current values from the current density maps across the three sites (columns) for four guilds (rows, colour-coded), including means and standard deviations. Current density values were log-transformed to improve visualisation of highly skewed data. The movement threshold (vertical dotted line) represents the threshold below which current flow is trivial (near-zero). A legend is to the right of the panels.

4.3.2 Site comparisons

The emerging urban site supported the most movement across the landscape, and existing functional connectivity was weakly supported between most protected areas. Current density maps showed relatively limited areas of near-zero movement, and continuous paths of low-moderate movement existed between protected areas for all three short-ranging guilds (Figure 27). Current histograms support this finding, showing that a greater proportion of cells supported movement than at the other two sites, with higher means and noticeably shorter tails extending below the movement threshold (Figure 29). Movement was mostly supported by vegetation on rural private blocks (west of the site) and a network of natively vegetated parks and swales through the suburban matrix (centre-east of the site). As a result, rehabilitation priorities in the site centre more around enhancing low-level connectivity, with this being the primary management objective recommended along key movement pathways (Figure 27). Patch emigration maps suggest a focus on enhancing connectivity between focal patches in the south-east of the site. Specifically, the two large habitat patches and surrounding stepping stones exhibited low proportions of successful emigration. With large perimeters abutting suburban development, most of their current leaked out within the matrix rather than reaching the scarce and small surrounding destination patches. Some functional connections existed between these patches, though they were generally weak and highly channelled along the swale network. This also indicates a need to improve network redundancy by rehabilitating the surrounding matrix and providing stepping stones along alternate pathways, as suggested by the framework's management actions.

Inversely, the existing urban and rural sites supported less overall movement across the landscape and featured prominent gaps in connectivity between protected areas. A much larger proportion of these sites had current values below the movement threshold and overall had lower mean currents, compared with the emerging urban site (Figure 29). Gaps in functional connectivity were also evident for all short-ranging guilds between protected areas at these two sites. At the existing urban site, a gap in movement probability was evident in the centre of the site, separating the protected areas to the north and south (Figure 26). While isolated stepping stones existed through the matrix, particularly in the east of the site for both woodland birds and terrestrial fauna, they were not close enough to enable functional connectivity. Patch emigration maps echoed this by highlighting the relatively high isolation of these intermediate stepping stone patches, particularly for terrestrial fauna.

At the rural site, protected areas were distanced well beyond the dispersal threshold of all short-ranging guilds, with the intermediate farmland providing limited to no stepping stones to support movement between them (Figure 28). For connectivity between protected areas to the west and the east of the site, the most feasible rehabilitation opportunity would incorporate the central protected area to reduce the need for establishing large stepping stones across a wide matrix. At both the existing urban and rural sites, candidate areas suggested that stepping stones are needed to effectively re-establish connectivity between protected areas (Figure 26, Figure 28). The gap in functional connectivity was larger at the rural site, however, the suitable areas for establishing adequate stepping stones were widespread and continuous. The homogenous farmland provided broader rehabilitation-friendly landscapes than the spatially constrained greenspaces of the urban environment.

4.3.3 Guild comparisons

Focal patch area for woodland birds and terrestrial fauna had a large degree of overlap across all three sites (98.8%) (Figure 30). Only on a few occasions did patches with enough structural shrub cover and diversity to be considered a focal patch for woodland birds exist without a groundcover layer. These were primarily in the south-east of both the existing urban and rural sites (Figure 26, Figure 28). Their movement abilities were also similar, with current values between the two guilds showing a very strong correlation across all sites (rho = 0.85) (Figure 31). However, terrestrial fauna were more strongly affected by barriers, which in some cases reduced their dispersal distance (Figure 32a) and in others created more channelisation of movement into low-resistance routes – Figure 32b shows how terrestrial fauna movement probability was more strongly channelled along the vegetated swale network in the emerging urban site. In some instances, this difference caused patch emigration success to be lower in terrestrial fauna for spatially similar or identical focal patches. For example, the cluster of focal patches to the south-east of the existing urban site (Figure 26), or the protected area patches in the south-west of the emerging urban site (Figure 27). These differences could alter rehabilitation priorities at finer scales, to ensure movement is adequately improved for terrestrial fauna around barriers and between patches that are more functionally isolated.

Wetland fauna showed slightly more constrained movement, generally due to less suitable habitat within the landscape. The guild shared approximately 41% of focal patch area with both woodland birds and terrestrial fauna, reflecting their more distinctive ecological niche (Figure 30). Current was more correlated with terrestrial fauna (rho = 0.62) than with woodland birds (rho = 0.52) (Figure 31), likely due to their more similar movement abilities and sensitivity to barriers which informed their resistance layers. The restricted movement of wetland fauna was most pronounced in the existing urban site, which contained substantially less focal patch area than the other guilds and larger gaps in functional connectivity, reflected in the current density maps, patch emigration maps (Figure 26) and current histograms (Figure 29). To bridge this gap between highly isolated habitats, candidate areas suggested more intensive rehabilitation work is required to create new stepping stone habitat. At the other two sites, however, differences were more subtle. The emerging urban site was built on a low-lying area, and many urban parks were damp or inundated, providing suitable habitat for wetland fauna as well as the other two short-ranging guilds. Most rehabilitation candidate areas aligned with those for woodland birds and terrestrial fauna, but a few additional stepping stones were identified for wetland fauna where they were absent for the other two guilds (Figure 27). Some of these differences reflected how current flowed within larger remnants, where multiple wetland focal patches occurred in close proximity. In these areas, current tended to concentrate within the remnant rather than flow towards stepping stones in the higher-resistance matrix.

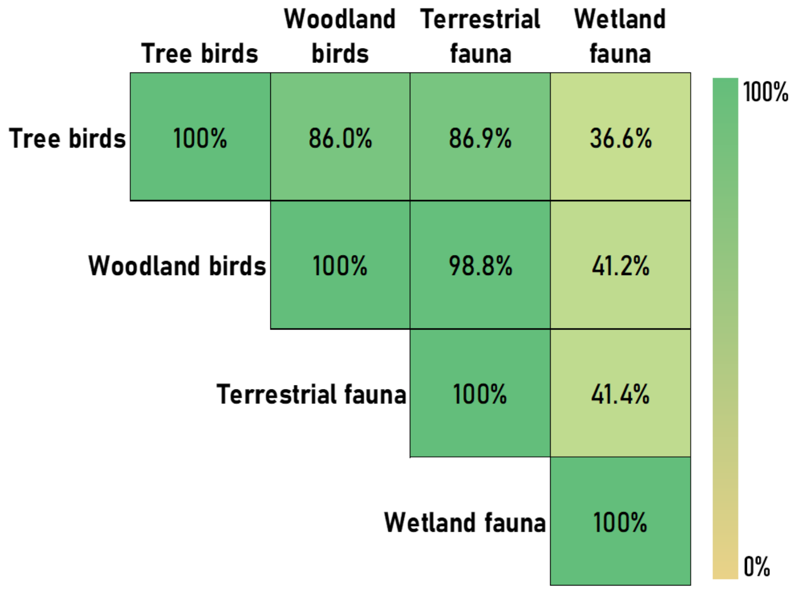


Figure 30. Pairwise matrix showing the spatial overlap of focal patches between the four ecological guilds (Jaccard similarity index, intersection area divided by union area). Overlap percentage is shown on a colour gradient from yellow (low) to green (high).

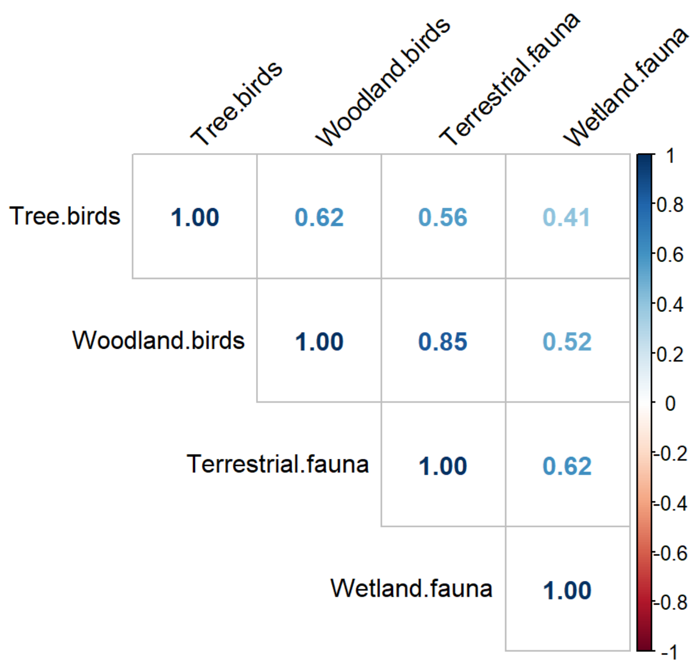


Figure 31. Pairwise matrix showing the Spearman correlations of current density values at all sites for the four ecological guilds. Spearman correlations are shown on a gradient from red (negative correlation) to white (no correlation) to blue (positive correlation).

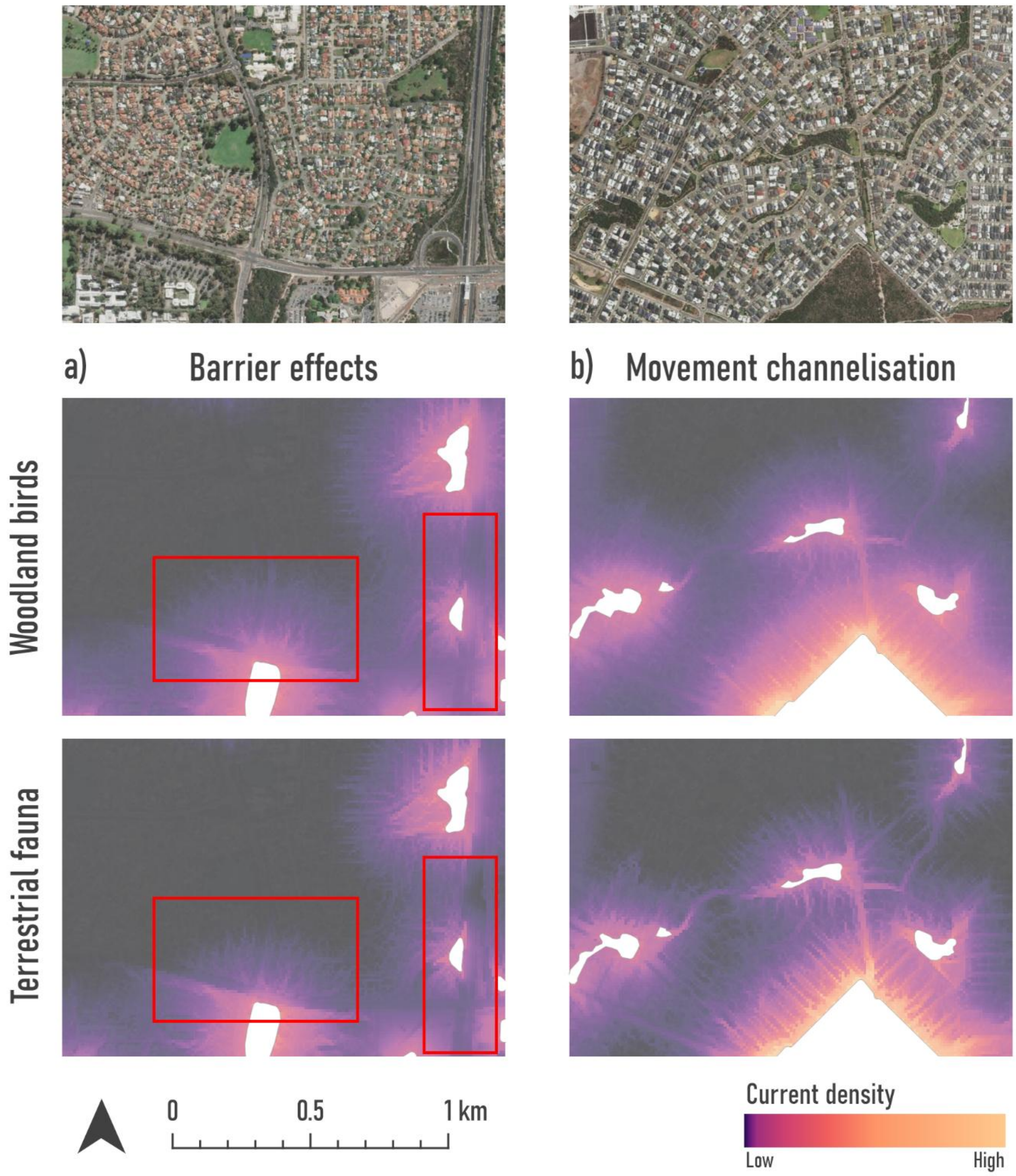


Figure 32. Comparison examples of current density maps showing a) the effect of barriers, and b) movement channelisation, for two guilds (woodland birds, middle row; and terrestrial fauna, bottom row). A satellite image is provided in the top row for context (Imagery © 2026 Google, Airbus, CNES / Airbus, Maxar Technologies, Vexcel Imaging US). Movement probability is shown along a colour gradient from dark (low movement) to bright (high movement) using a consistent colour stretch; a legend is provided below the panels. Red squares indicate comparison areas, showing where barriers more strongly reduce movement for terrestrial fauna.

4.3.4 From connectivity patterns to rehabilitation priorities

Overall, the rehabilitation framework and outputs support the identification of rehabilitation priorities for the three short-ranging guilds. The current density values varied broadly (Figure 29), while the maps revealed clear patterns in movement potential, ranging from high probability areas to extensive areas of near-zero movement (Figure 26-Figure 28). Dispersal limitations were particularly evident, with varying proportions of the landscape falling below the movement threshold (Figure 29) and some patches exhibiting extremely low emigration success (as low as 0.03%). These areas of functional disconnection located a range of candidate areas that were potentially suitable for establishing stepping stone habitat (Figure 26-Figure 28).

Where movement was present, current density maps highlighted both high-movement corridors and more diffuse or channelled low-movement pathways. High movement routes were most apparent between nearby focal patches, such as in the south-west of the existing urban site (Figure 26) and the west of the emerging urban site (Figure 27). Degraded high-flow areas were also highlighted in candidate area maps, but were quite sparse and often limited to areas immediately surrounding large focal patches (Figure 26-Figure 28). This pattern reflects how current increases exponentially approaching a focal patch, and this proximity effect can overwhelm more subtle landscape-driven differences in movement potential. As a result, the single threshold used to delineate high-flow areas may capture proximity effects more than landscape differences, particularly around large habitat patches with very high local currents. Therefore, identifying high-flow areas should also draw on current density maps, which provide finer-scale gradients that can help distinguish meaningful high-flow routes near focal patches.

Low movement routes often included diffuse current over homogenous landscapes, most typical in the rural site (Figure 28); and ridges of current along corridors of relatively low resistance, illustrated best by the swale network in the emerging urban site (Figure 27). These were often coupled with patches showing low emigration success, which provided more details as to which patches require additional support. Candidate areas for enhancing low-level connectivity were broad, but they clarified where matrix improvements alone were sufficient and where movement had declined to the point that new stepping stone habitat was required. The candidate area maps effectively indicated the transition along key movement pathways from zones where matrix enhancement could maintain movement, to zones where new stepping stones became necessary to sustain functional connectivity.

4.3.5 Unexpected results for long-ranging guilds

The tree birds guild produced results that made it difficult to interpret rehabilitation priorities. Artefacts appeared in the outputs that seem to reflect a misalignment between the movement ecology of longer-ranging species and the assumptions of the one-to-all Circuitscape configuration used. Current values for this guild never dropped below the movement threshold (Figure 29), and the successful emigration rates were consistently high (Figure 33), suggesting functional connectivity across the entire sites. However, the one-to-all approach assumes that organisms will settle (i.e. to go to ground) at nearby destinations rather than continue to more distant patches. This assumption was practical, and even suitable, for the shorter-ranging guilds where dispersal limitations were a primary feature shaping movement patterns (Figure 26-Figure 28). However, for tree birds, this assumption created a noticeable ‘shadow’ of low movement predictions beyond nearby destination patches, evident at all three sites (Figure 33). In the existing urban site, current left large source patches within the protected areas and travelled to nearby small destination patches, after which current reduced drastically, leaving a dark area of low current in the centre of the study area. A similar dark area is seen in the emerging urban site around a cluster of small focal patches that absorbed outgoing current. In the rural site, current emitted from the western protected area decreased rapidly after hitting small nearby destination patches, and travelled further east in the absence of those patches. Overall, this hindered the accurate prediction of movement for the longer-ranging guild and obscured rehabilitation priorities.

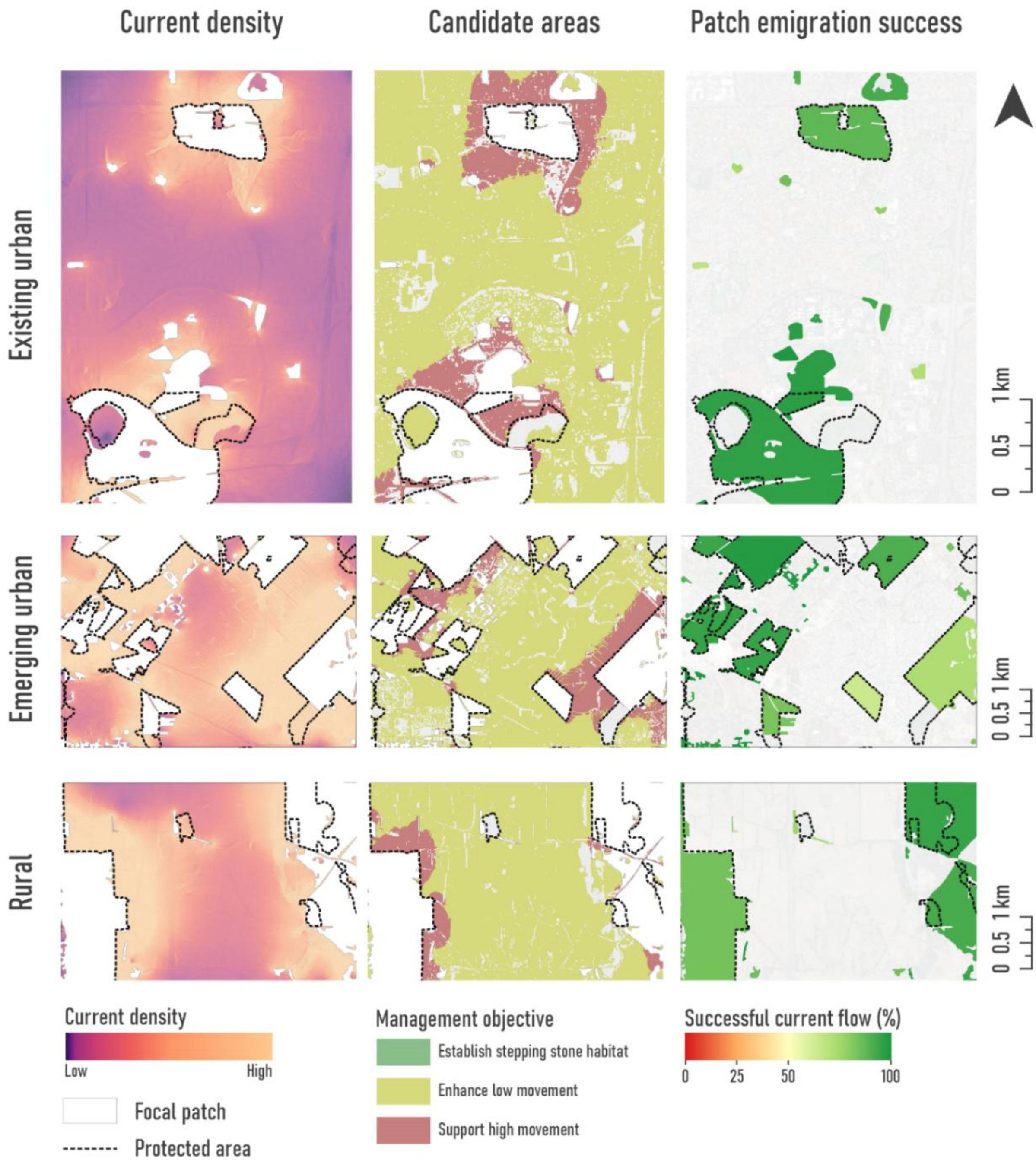


Figure 33. Outputs of the connectivity analyses for the tree birds guild at all three sites. Maps show (columns) current density, rehabilitation candidate areas, and patch emigration success, presented for (rows) existing urban, emerging urban, and rural sites. Current density maps illustrate the distribution of movement probability. Rehabilitation candidate maps provide management objectives for all cells based on movement and underlying landscape resistance. Patch emigration success maps show the proportion of current emitted from each patch that successfully reaches another patch. Legends for each map type are provided below, with consistent colour stretches used across panels.

4.3.6 Rehabilitation demonstration cases

The outputs of this study offer an explicit interpretive framework for identifying rehabilitation priorities based on current movement patterns at the local scale. Importantly, exact rehabilitation priorities at any site are not absolute and depend on the management goals, target guild, stakeholder preferences, and constraints of implementation.

Nonetheless, with the goal of increasing successful movement along the high-priority linkages identified in Chapter 3 and balancing outcomes among short-ranging guilds, several broad recommendations can be made. Below are demonstrative examples of rehabilitation priorities at each site, with locations summarised in Figure 34-Figure 36. Each rehabilitation area is assigned a management objective based on the dominant candidate area patterns within the simplified polygons, supported by cumulative current maps, especially for delineating high-flow corridors. Where rehabilitation areas are guild-specific, detail is provided in the accompanying text. The assigned objectives are illustrative of the general approach rather than prescriptive, with further explanation is provided below.

In the existing urban site, rehabilitation priorities include (Figure 34):

a) utilising pathways of intermediate habitat on the eastern side of the site to form the foundation of the linkage;

b) creating stepping stone habitats in the north-east quarter of the site to re-establish movement between isolated patches;

c) matrix enhancements to support movement along key pathways, with a focus on woodland birds and terrestrial fauna in the eastern recommendation areas where wetland fauna habitat is absent; and

d) rehabilitating degraded areas and easing barriers where possible along existing high movement corridors in the southern half of the site, and along future movement pathways to reduce potential harm.

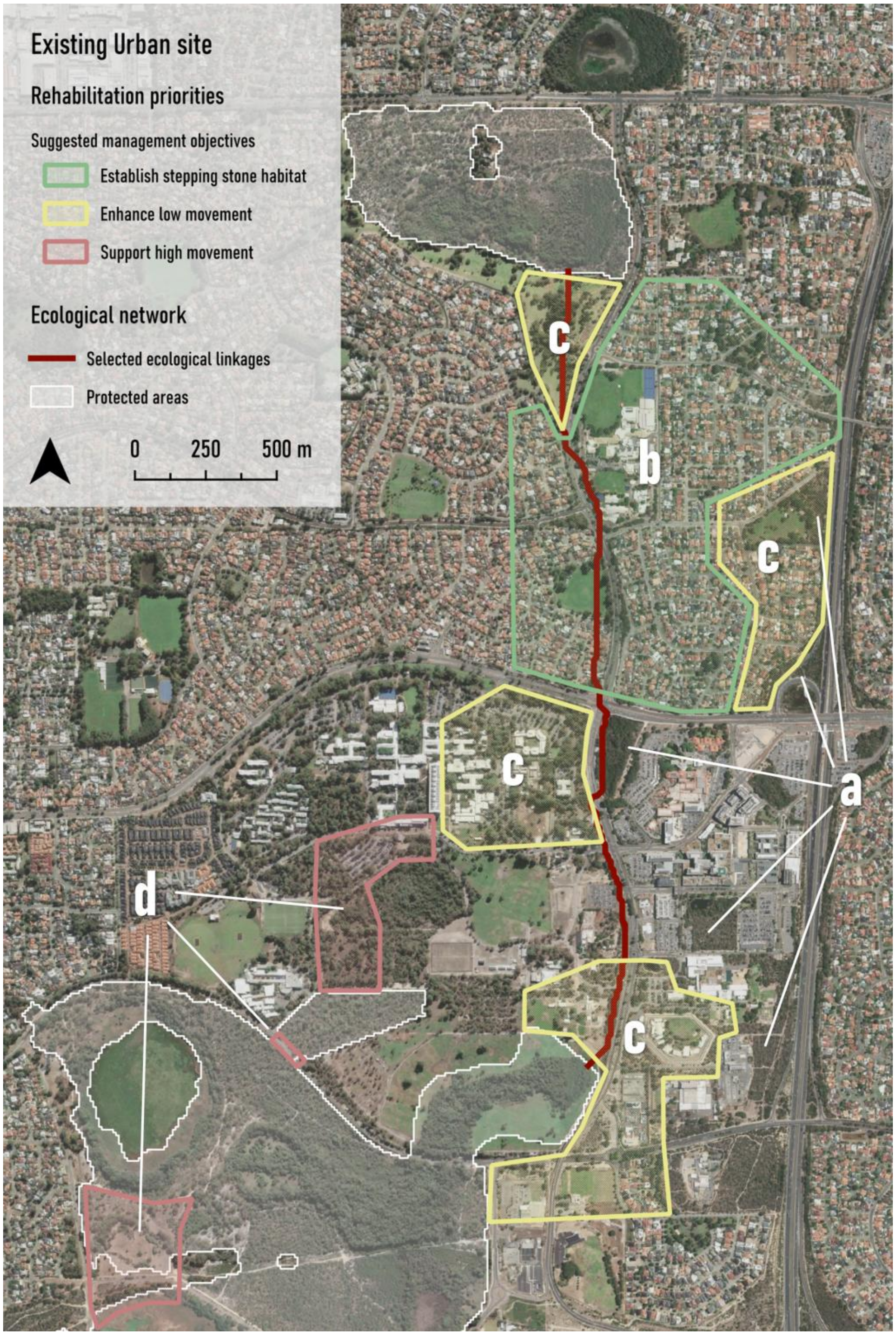


Figure 34. Recommendations for rehabilitation priorities at the existing urban site, with the goal of enhancing connectivity between protected areas. Maps show the location of suggested rehabilitation areas and broad management approach. Letters (a, b, c, d) relate to the list of rehabilitation priorities at each site, described in the thesis text. Protected areas are shown in white, while the high-priority linkages selected as case studies for local-scale prioritisations are shown in dark red. Base maps: Imagery ©2026 Google, Airbus, CNES / Airbus, Maxar Technologies, Vexcel Imaging US.

In the emerging urban site, rehabilitation priorities include (Figure 35):

a) enhancing movement along the existing ecological linkages at key locations, including establishing stepping stones where predicted movement is very low (particularly for wetland fauna) and easing the matrix along degraded, moderate-high flow corridors;

b) enhancing route redundancy between protected areas in the south-east of the site by improving matrix permeability and establishing small, targeted stepping stones where current goes to ground;

c) supporting the existing channelised low movement pathways by enhancing habitat quality along swales, establishing targeted stepping stones, and increasing redundancy by establishing alternative pathways in the surrounding matrix; and

d) enhancing matrix permeability in diffuse flow zones in the site's north-east.

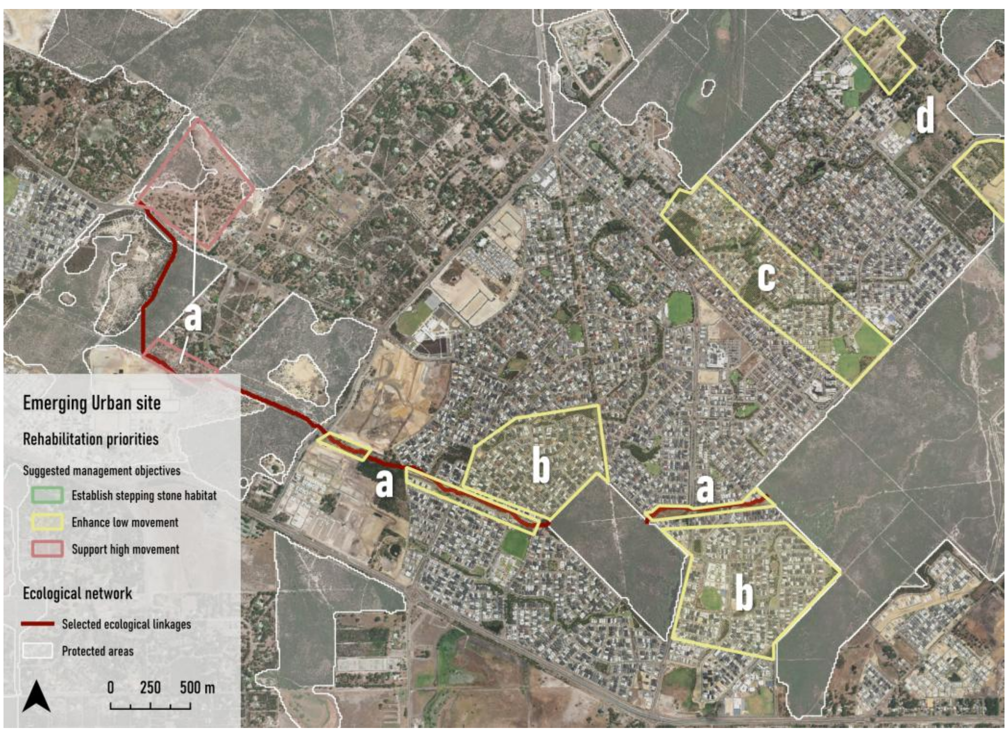


Figure 35. Recommendations for rehabilitation priorities at the emerging urban site, with the goal of enhancing connectivity between protected areas. Maps show the location of suggested rehabilitation areas and broad management approach. Letters (a, b, c) relate to the list of rehabilitation priorities at each site, described in the thesis text. Protected areas are shown in white, while the high-priority linkages selected as case studies for local-scale prioritisations are shown in dark red. Base maps: Imagery ©2026 Google, Airbus, CNES / Airbus, Maxar Technologies, Vexcel Imaging US.

In the rural site, rehabilitation priorities include (Figure 36):

a) creating strategically placed, high-quality stepping stone habitats across the narrower gaps between protected areas, supported by corridors or additional smaller stepping stones;

b) utilising, and rehabilitating around, the central protected area as an intermediate habitat to ensure the best chance of successful movement between core habitats on the west and east; and

c) rehabilitating the degraded section of the eastern linkage that exhibited diffuse movement.

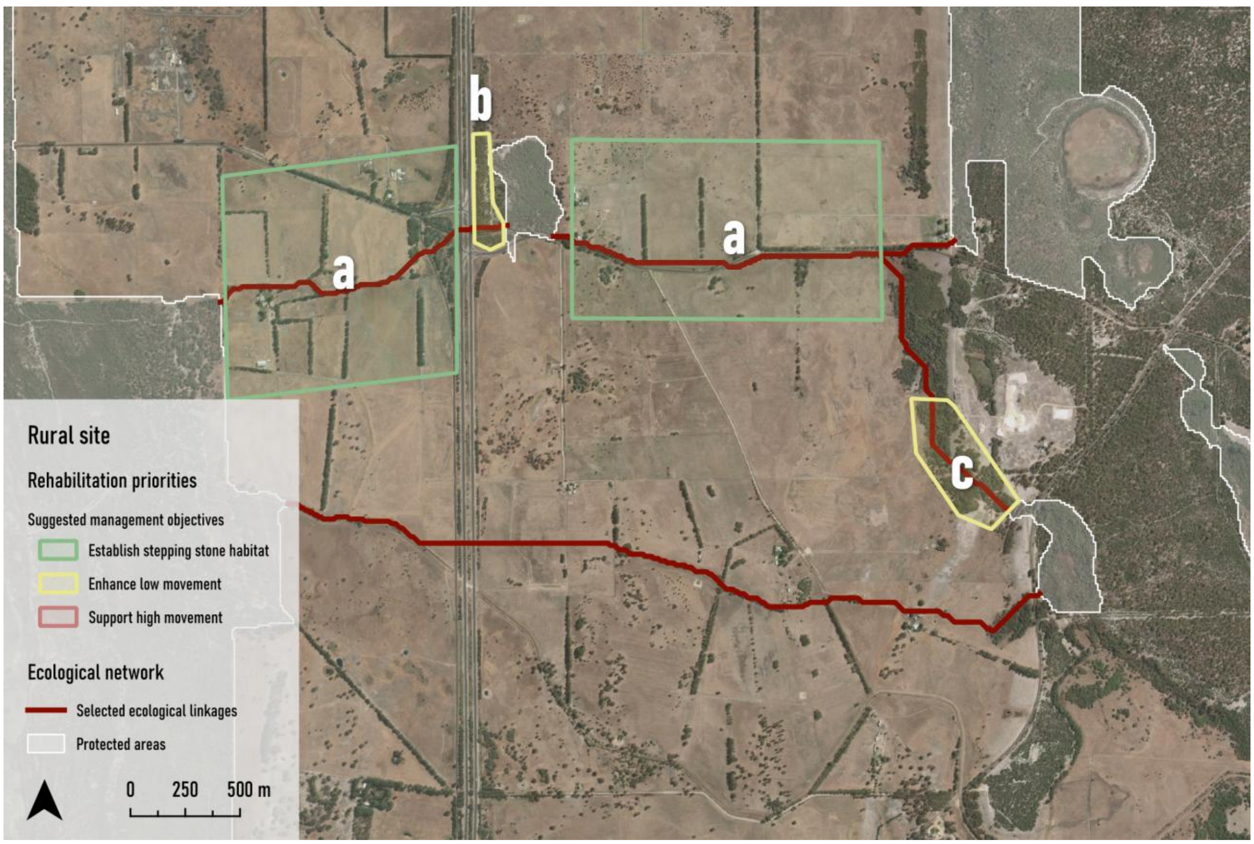


Figure 36. Recommendations for rehabilitation priorities at the rural site, with the goal of enhancing connectivity between protected areas. Maps show the location of suggested rehabilitation areas and broad management approach. Letters (a, b, c) relate to the list of rehabilitation priorities at each site, described in the thesis text. Protected areas are shown in white, while the high-priority linkages selected as case studies for local-scale prioritisations are shown in dark red. Base maps: Imagery ©2026 Google, Airbus, CNES / Airbus, Maxar Technologies, Vexcel Imaging US.

4.4 Discussion

4.4.1 Identifying the need for stepping stones to facilitate dispersal

In fragmented urban landscapes, effective connectivity improvement requires targeted interventions informed by both fine-scale landscape structure and the movement ecology of relevant species (Faeth *et al.*, 2011; Lynch, 2018; Donati *et al.*, 2022). This study provided insights into these functional connectivity patterns for multiple wildlife guilds at a local scale, enabling an understanding of how they influence on-ground rehabilitation priorities to enhance connectivity.

By incorporating dispersal limitations into predictive movement models, the framework echoes a fundamental structural hierarchy in ecological networks – that of core habitat, corridors and stepping stones, and matrix permeability. While core habitat is the backbone of ecological networks, providing essential stable resources for survival and reproduction (Hercock, 1997; Soga *et al.*, 2014), corridors and stepping stones are the primary mechanisms supporting connections between them; while these, in turn, are supported by matrix permeability (Baum *et al.*, 2004; Saura *et al.*, 2014). Although stepping stones were intentionally included as focal patches in the Circuitscape models, the outputs nonetheless illustrated the importance of intermediary habitat in enabling extended dispersal, particularly showcasing just how limited movement can be relative to the distances between protected areas commonly found in the urban landscape. The models also incorporate the role of matrix permeability in supporting dispersal and enhancing the effectiveness of stepping stones (Baum *et al.*, 2004), which was most evidenced in the emerging urban site through the vegetated swale network.

Unsurprisingly, these insights give the methodology potential beyond identifying rehabilitation priorities and can inform protection and conservation of crucial intermediary habitat patches. Though given much Circuitscape research is already focused on conservation (Dickson *et al.*, 2019), the insights potentially offer more value to rehabilitation planning. They add to the compounding literature illustrating that in many cases, rehabilitation of high-quality intermediary habitat is not only an option, but a necessity, for enabling connectivity of short-ranging species in human-modified landscapes (Saura *et al.*, 2014; Lynch, 2018; Ottewell *et al.*, 2019; Donati *et al.*, 2022). The framework illustrates where this necessity arises as current goes to ground, and provides management recommendations to prioritise strategically placed stepping stones or corridors, supported by enhanced matrix permeability.

4.4.2 Framework use

The three outputs – current density maps, focal patch-level emigration success, and set management recommendations – allow movement probabilities to be operationalised into systematic priorities with transparency, repeatability, and clarity for planning. Providing stakeholders with an evidence-based tool alongside interpretive guidance may help bridge the gap between connectivity research and implementation, as well as focus resources to enhance biodiversity outcomes (Keeley *et al.*, 2019). The findings of the framework also highlight broader considerations for how it can be used in practice.

Comparing guilds, the movement and habitat of terrestrial fauna and woodland birds were surprisingly similar. This was unexpected, as these guilds tend to inhabit different vegetation strata (Appendix B), which was accounted for when identifying focal patches. However, the results showed that in most cases, vegetation that provided for one tended to provide for the other, with intact native midstorey and understorey layers co-existing more often than not. Additionally, their movement patterns were unexpectedly similar, despite the dispersal advantages of flight for woodland birds. While the dispersal thresholds were set at similar distances for both guilds, the effects of barriers were set much higher for terrestrial fauna. Sensitivity testing was undertaken in setting resistances, which were increased twice for barriers for terrestrial fauna, but this didn’t create the expected pronounced difference between guilds. In some areas, terrestrial fauna models still showed current flow that traversed fine-scale landscape barriers (e.g. fences, road sound barriers, or topography changes) particularly in the existing urban or emerging urban sites. These features were simply not included in models due to heavy data requirements and the modelling complexities they would introduce. There is potential to integrate this data into connectivity models, but a balance should be made between model accuracy and practical resource demands that may hinder the framework's strength as a rapid and repeatable tool. A practical solution is that connectivity planning for these two guilds could be managed together, with on-ground interventions differing at the design stage. Informed decisions involving ecologists should be made during this process, however, to ensure the needs of the unique guilds are supported. For example, low fences, road barriers, or fine-scale topography are likely to be more consequential to terrestrial fauna movement than woodland birds (How and Dell, 2000; Shepard *et al.*, 2008). These aspects should be considered both when interpreting priorities and when managing actual implementation.

In alignment with Chapter 3, this study grounded rehabilitation priorities in the goal of enhancing connectivity between protected areas. However, the framework is flexible enough to discern other patterns from the outputs to be applied to other goals or management objectives. Connecting extremely isolated patches may be undesirable if it fails to create functional connections, and instead risks drawing organisms into ecological traps (Zuñiga-Palacios *et al.*, 2021), particularly if closer opportunities exist. In these cases, rehabilitation may instead focus on increasing connectivity between nearby patches or expanding isolated patches to improve habitat availability and population stability (Hanski, 1999; Faeth *et al.*, 2011). The framework's applicability remains when interpreting priorities for these objectives, or for others, by providing useful information about the degree of patch isolation, or predicted movement patterns towards nearer habitat clusters to which connectivity could be strengthened.

However, the one-to-all Circuitscape approach does not provide a simple and accurate measure of pairwise connectivity (McRae *et al.*, 2013). While it provides valuable information for planning targeted interventions, it is not suitable for quantitative comparisons of connectivity between specific habitat patches before and after rehabilitation. This places the framework less as an evaluative tool in this sense, though it may be suitable for comparing spatial current patterns and overall landscape movement before and after rehabilitation. While within the aims of this study, it is important to note that the outputs cannot be extrapolated for pairwise evaluations.

4.4.3 Limitations and use considerations

Candidate area thresholds based on current flow were unable to control for the combined effects of distance from focal patches and variation in current emitted by patches of different sizes. The exponential rise in current nearing a focal patch sometimes obscured the effects of landscape composition on predicted movement. As a result, candidate areas often showed broad areas for rehabilitation that reduced the precision of recommendations. Despite this, incorporating dispersal limitations was essential to capturing realistic functional connectivity. Circuitscape models incorporate complex interactions between multiple movement pathways, habitat size and composition and landscape resistance – detail unable to be replicated by focal patch buffers alone. An attempt was made to statistically remove the effects of patch size and proximity by modelling current as a function of these variables and extracting residuals. However, the approach did not meaningfully increase interpretive value of outputs without adding unnecessary analytical complexity. Therefore, the original model outputs were accepted with these limitations. Future research could explore methods for circuit theory to integrate dispersal limited models with means to isolate the effects of landscape composition.

Some limitations were also introduced by the model assumption that organisms will settle at nearby destination patches rather than continue through the matrix. This assumption was accepted as it generally aligned with the movement ecology of often dispersal-limited short-ranging fauna (How and Dell, 2000; Bush *et al.*, 2010; Ottewell *et al.*, 2019). For the tree birds guild, however, representative species have relatively strong dispersal abilities and their movements are driven by resource availability, often following paths of fragmented native vegetation (Rycken, 2019), making this assumption unsuitable. Alternative Circuitscape configurations using pairwise approaches applied over larger spatial extents may better capture pinch points and diffuse flow zones, improving the identification of rehabilitation priorities for these species. For wetland fauna, multiple focal patches often existed in a larger vegetation remnant, causing current to gravitate between each other rather than traverse the matrix. This effect is likely to be ecologically realistic as the increased shelter and moisture provides preferable movement pathways (Bush *et al.*, 2010; Santoro *et al.*, 2020). Wetland fauna movement patterns and rehabilitation recommendations are likely not to be internally affected, though this will affect comparisons with the other two guilds, which assume perfect connectivity within a larger vegetation remnant encompassed by a single focal patch. The assumption also affects the calculation of patch emigration success. This metric identified patches isolated from all other patches, though it amplified the emigration success of clusters of patches that were nearby to each other, but far from others. This could cause legitimately isolated patches to be missed.

A final note relates to the inherent assumptions and uncertainty embedded in all connectivity models (Riordan-Short *et al.*, 2023). While decisions were informed by research, sensitivity testing, and structured field-based refinements, the models are imperfect due to the complex nature of movement ecology the study aims to recreate. A few suggestions for future research may enhance the reliability and enable more accurate interpretation of priorities:

- The field refinement process, though systematic, still introduces some subjectivity. More structured metrics for defining and measuring ‘habitat’ and ‘stepping stones’ would help standardise assessments of urban connectivity. While the Keighery vegetation condition scale widely used for assessing native vegetation condition in Western Australia (Keighery, 1994; EPA, 2016), it is less suited to assessing habitat quality in novel or hybrid urban landscapes. New habitat quality metrics that account for ecological novelty would improve consistency and reliability in urban ecology, and may be defined at community, guild or species levels.

- Limited movement data was available for local fauna species. Further empirical research on the movement ecology of Australian species is likely to significantly improve connectivity models in the region.

- This study used a CHM to characterise vegetation structure, however, this created large uncertainties for understorey vegetation which were not captured under dense canopy. Among many other applications, developing remote sensing methods to assess vertical vegetation structure below canopy would be valuable for connectivity modelling and reduce field validation requirements.

- Connectivity models are more robust where outputs are validated with independent data (Riordan-Short *et al.*, 2023). While this study did not do this under time limitations, validation of model outputs with species occurrence or telemetry data could help to further verify model success.

Conclusion: a methodological foundation for circuit theory-based rehabilitation priorities

This study demonstrates that a dispersal-based, guild-specific local connectivity framework can translate complex patterns in movement probability into spatial and managerial rehabilitation priorities. Integrating current density maps, patch emigration success and candidate rehabilitation areas in a transparent framework, the study systematically identifies where interventions such as stepping stones, matrix enhancement, barrier easement, or enhancing redundancy can enhance functional connectivity for a range of species. While the models carry inherent limitations – relating to proximity and patch size effects, or guild-specific movement assumptions – the framework reflects structural realities of ecological networks and captures the inherent worth of stepping stones and matrix permeability for the dispersal of short-ranged fauna (Baum *et al.*, 2004; Saura *et al.*, 2014; Lynch, 2018). The outputs remain flexible to support a range of management objectives and connectivity goals, also highlighting the importance of existing habitat patches for conservation. Overall, the framework provides a useful decision support tool for conservation and planning practitioners seeking to prioritise rehabilitation in human-modified landscapes. The study establishes a methodological foundation for translating functional connectivity patterns into actionable rehabilitation interventions.

CHAPTER 5:

GENERAL DISCUSSION

This thesis developed a framework that can spatially prioritise rehabilitation at regional and local scales, to enhance connectivity for a range of wildlife species. The Perth-Peel region is a highly heterogenous landscape, with an ecological network that spans from highly urbanised landscapes to swathes of native forest. The thesis uncovered the potential roles these different parts of the city exhibit when seeking to enhance the ecological network, and ranked which linkages provide the best capacity to maximise biodiversity outcomes in the region. It also provided a local-scale framework to map functional connectivity and translate these regional priorities into targeted, on-ground rehabilitation priorities for multiple wildlife guilds. Together, these investigations offer insights on where to invest in rehabilitation, and how this planning can be used to benefit a range of species. They provide transparent and adaptable decision-support tools to guide practitioners in rehabilitation planning to maximise connectivity outcomes.

5.1 Where to invest in rehabilitation

Rehabilitation for connectivity is not best placed in areas of high biodiversity value alone. More pragmatically, the best opportunities exist where the landscape is fragmented but not to extremes, making functional connectivity and rehabilitation *necessary* but also *feasible*. Variation in regional priorities was most driven by linkage characteristics such as length, permeability, or distances between remnant habitats, rather than protected area characteristics (conservation values or vulnerability), indicating that the landscape context dominated the outcomes, not the need for conservation. The dominance of linkage structure in influencing overall priorities was particularly evident in urban areas, in which there were very few high-priority linkages, while it was less pronounced in peri-urban and rural areas where high-priority linkages were more common. These findings highlight the importance of enhancing the few urban linkages where high biodiversity and vulnerability coincide with structural opportunities for meaningful connectivity gains between otherwise isolated protected areas. These linkages represent rare and valuable opportunities for urban conservation, which can be crucial to maintaining threatened species in urban environments (Soanes and Lentini, 2019). Meanwhile, planners could take advantage of vaster rural opportunities to improve whole-network connectivity and redundancy. This is likely to be reflected in other cities, considering urban edge-to-centre biodiversity gradients predict higher biodiversity in outer-urban regions (McDonnell and Hahs, 2008), suggesting that strategic network stability could be achieved through broad rural linkage rehabilitation.

The local case studies reiterated how linkage structure facilitates meaningful connectivity gains, illustrating how stepping stones and supporting matrix permeability are necessary to extend organism dispersal (Saura *et al.*, 2014; Lynch, 2018). This was most evident in the emerging urban site, where wetland habitats and vegetated swales enabled functional connections through the suburban environment for all three short-ranging guilds. Rehabilitation priorities modestly focused on strengthening these paths and increasing redundancy where channelled movement created connectivity bottlenecks. Conversely, the existing urban and rural sites lacked intermediate habitats, severing functional connectivity and shifting priorities to establishing new stepping stones to bridge this disconnection. In the context of the model assumptions, re-establishing connectivity requires habitat to be of adequate quality to support temporary settlement between multiple dispersal events. This may amplify rehabilitation effort, requiring more intensive interventions and maintenance to establish suitable and structurally complex plant communities (Wilson *et al.*, 2012; Brancalion *et al.*, 2019), which can greatly increase costs when connectivity is severed over longer distances. Feasibility may be further constrained by major barriers (Shepard *et al.*, 2008), or where suitable public land for new stepping stones is limited, potentially necessitating more expensive options such as land acquisition, easements, or use of exceedingly modified sites. Overall, while increasing matrix permeability is most effective in landscapes with existing functional connectivity, it plays a supportive role relative to intermediary habitats (Baum *et al.*, 2004). Where these habitats are absent, new stepping stones (or corridors) remain the primary means of addressing functional disconnection, with their placement shaped by both the extent of the disconnection and the practical constraints of the surrounding landscape.

Together, the regional and local investigations revealed spatial leverage points that neither alone would have identified. The MCDA efficiently narrowed the network to a manageable set of high-opportunity linkages, and local models revealed detailed movement gradients across the entire linkage, enabling small, targeted interventions to unlock functional connectivity between protected areas for a range of species. In practice, this targeted investment is likely to provide higher biodiversity returns per unit effort than uncoordinated greening or the use of a single planning scale (Patterson *et al.*, 2023; Croeser *et al.*, 2024).

5.2 Multi-guild wildlife planning

Local-scale analyses revealed potential wildlife guild responses to landscape structure, identifying that while connectivity planning can be co-managed for some guilds, others will require bespoke approaches. Woodland birds and terrestrial fauna models showed marked similarities in habitat and dispersal across the three case study sites, meaning high-priority linkages and local rehabilitation priorities are likely to be similar for both guilds. What will primarily differ are design-stage interventions, requiring due consideration of the different guild movement modalities. For instance, barriers such as roads, fences, or fine-scale topography changes are more likely to affect ground-dwelling than avian fauna (Garden *et al.*, 2006).

The wetland fauna model, however, was strongly constrained by habitat isolation, particularly in the existing urban case study where potential stepping stone wetlands were scarce and often degraded. The vegetated wetlands and swales in the emerging urban site conversely provided intermediate habitat which helped to bridge functional connectivity gaps. For this guild, improving connectivity is likely to require targeted rehabilitation of degraded blue infrastructure to increase habitat availability and reduce isolation between more ecologically supportive systems.

These guild-specific patterns highlight why regional priorities must be interpreted with care. At broad scales, the framework necessarily relied on generalisations across species, using dispersal thresholds from a range of local species to characterise ideal linkage lengths, or assuming barriers were of equal consequence across all guilds. However, while short urban linkages without barriers may be particularly valuable for ground-dwelling fauna, they may be unnecessary for highly mobile avian species (How and Dell, 2000; Shepard *et al.*, 2008; Rycken *et al.*, 2022). This underscores the need to inspect individual linkage details to best understand how it may present opportunities and challenges for specific guilds of interest in a given area or for relevant stakeholders.

5.3 Planning implications

This framework utilises distinct analysis types that provide modularity for practical use. Tied together by the ecological network, each analysis provides a tool with unique ecological information relevant to the different planning scales. The regional prioritisation provides key insights and policy relevance for state planning, specifically for enhancing Perth's 'Green Network', established in 'Directions 2031 and Beyond' (DPLH and WAPC, 2010) and continued in the 'Perth and Peel @3.5 million' and 'Urban Greening Strategy' frameworks (DPLH and WAPC, 2015, 2026). Meanwhile, the local models guide rehabilitation at the sub-linkage scale, typically managed by local government. In Perth-Peel, local governments manage biodiversity through local policy, maintenance of government land, and biodiversity incentives on private land (Pauli and Boruff, 2016). This scale is relevant for Local Biodiversity Strategy implementation, providing actionable guidance to local governments on which land parcels should be targeted, which management actions are required, and which species should guide the specific vegetation interventions. The modular framework could also be applied in other parts of the world, addressing similar planning and management entities at the two scales.

Practitioners are likely to need processes that help translate broad priorities into local actions that are feasible and stakeholder-aligned (Aronson *et al.*, 2017). Management goals and stakeholder preferences can vary, such that some may wish to target linkages of high investment and high return, while others may search for low stakes or 'easy wins'. As with guild-specific differences, linkage details should be consulted to determine which linkages are most suitable for a particular project before undertaking a local-scale assessment. Below is a practical workflow for applying the framework within the Perth-Peel region, or to be adapted in other regions of the world. The only step requiring specialised knowledge of Circuitscape modelling is Step 2, though the entire process would benefit from ecological expertise provided by consulting or academia to guide interpretation and implementation.

1. **Linkage screening:** Identify relevant high-priority linkages and interrogate linkage details against project goals.
2. **Local rehabilitation analysis:** Apply the local framework to map baseline functional connectivity and identify rehabilitation priorities.
3. **Rehabilitation and monitoring:** Design and implement rehabilitation in collaboration with relevant stakeholders, and track project success.

5.4 Future directions

Future research could build on this work in several ways. The regional prioritisation relied on existing ecological and spatial data, so incorporating socio-ecological perspectives through a participatory approach – engaging local planners and ecologists in criteria identification, threshold setting and weights – could ease uncertainties around practical challenges to ensure priorities align with local expertise and preferences (Krueger *et al.*, 2012; Sutherland and Burgman, 2015). Given the diverse goals of urban conservation (Threlfall *et al.*, 2019), the regional methodology can also apply to goals beyond rehabilitation, such as conserving high-functioning or valuable linkages, or supporting specific ecosystem services.

The local framework used a novel approach and carried limitations related to modelling complexity, proximity effects, and guild-specific movement assumptions. Future refinements could reduce modelling uncertainties to increase repeatability and practitioner accessibility; integrate dispersal limitations with approaches that isolate the effects of landscape composition, improving the selection of candidate rehabilitation areas; and model functional connectivity of longer-ranging species that were not well served by the current methodology. Alternative Circuitscape configurations for the tree birds guild could integrate and complement ongoing research on the movement, food preferences, and ecology of threatened black cockatoo species in Perth-Peel (e.g. Rycken, 2019; Rycken *et al.*, 2022; Riley *et al.*, 2023), further enhancing conservation planning for these species.

5.5 Conclusion: bridging connectivity theory and practice in a biodiversity hotspot

It has been long understood that functional connectivity is crucial for metapopulation stability in fragmented landscapes (Taylor *et al.*, 1993; Hanski, 1999), and this principle is increasingly relevant to modern urban conservation (LaPoint *et al.*, 2015; Soanes *et al.*, 2020, 2023). Despite Perth-Peel's location within a global biodiversity hotspot (Myers *et al.*, 2000), the region, like much of Australia, faces an implementation gap between connectivity theory and practice. Few planning tools are available to enhance connectivity (e.g. Kirk *et al.*, 2023; Norman and Mackey, 2024), especially integrating multiple scales and species.

By integrating methodological approaches at regional and local scales, this thesis provides insight into where to invest in rehabilitation and how to enhance connectivity for multiple wildlife guilds. It provides a set of repeatable decision support tools for practitioners who seek to enhance connectivity in urban environments, with each scale providing relevance to planning realities in Perth-Peel and wider Australia. The spatial outputs are freely available to be explored and inform green infrastructure planning in Perth and Peel (https://arcg.is/11bKGe1). While regional prioritisation results are likely to be accessible and understood by any organisation, collaboration with ecologists from consultancies or academia is recommended, especially for the local scale framework, to perform ecological modelling and interpret rehabilitation priorities in specific environmental contexts. It is hoped that this research acts as a methodological foundation to evolve Perth-Peel's regional ecological network into strategic action for network repair, providing inspiration to others across the globe to do the same.

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APPENDIX A – DETAILED CRITERIA CALCULATIONS

Vegetation complex rarity

The *South West Vegetation Complex Statistics* dataset (DBCA, 2019) lists the 'percent remaining' of each vegetation complex as compared to its pre-European extent. To summarise vegetation rarity within each protected area, an area-weighted average was calculated of the 'percent remaining' of vegetation complexes present within a protected area. Two spatial datasets outlining the locations of these vegetation complexes – covering the Swan Coastal Plain (DBCA, 2018b) and South West Forest Region (DBCA, 2018a) – were merged to ensure full coverage of the study area. Because these datasets represent vegetation complexes tied to specific soil and landscape patterns, they are not limited to remnant vegetation but also cover the areas that have been cleared. The weighted 'percent remaining' of vegetation complexes was calculated using the formula:

$$Weighted\ percent\ remaining = \sum_{i=1}^{n} P_i \times R_i$$

Where P_i is the proportion of the protected area occupied by vegetation complex *i*, and R_i is the 'percent remaining' of vegetation complex *i* within the broader region. Since the sums equal to 1, no denominator is required.

Threatened and priority ecological community (TEC/PEC) extent & diversity

A threatened species and communities database search was requested on the 31st March 2025 (DBCA, 2025c). A unique index, *TEC/PEC extent and diversity,* was calculated for each protected area to indicate TEC/PEC values. This index was based on the total area of TECs and PECs within the protected area (to represent extent), with a 20% multiplier for each additional unique community type present (representing diversity):

$$TEC/PEC\ extent\ and\ diversity = P \times (1 + 0.2(n - 1))$$

Where P is the proportion of the protected area that is covered by TECs or PECs, and n is the number of unique community types.

Threatened and priority flora and fauna richness

The *DBCA threatened species dataset* (DBCA, 2025c) was first filtered to ensure records were current and relevant to the analysis. Records were considered relevant if they were less than 40 years old, not cultivated if flora, and not marine if fauna. Species richness was calculated separately for T&P flora and fauna within each protected area, including a buffer to account for slight spatial inaccuracies and species movement. Flora richness was calculated with a buffer of 100m, while fauna richness was calculated with a buffer of 500m given their greater dispersal ability.

Spatial sampling bias was corrected for each T&P flora and fauna, using log-log linear models of T&P species richness against a proxy for survey effort. Survey effort was estimated for each protected area using the total number of records from Dandjoo (the primary repository of spatial biodiversity data in Western Australia, combining data from government, industry, research and community) and *DBCA threatened species dataset* (DBCA, 2025a, 2025c), representing the total records of T&P and non-T&P species. Using available attributes, Dandjoo records were filtered using the same relevancy constraints as the *DBCA threatened species dataset* to ensure data comparability. Additionally, Dandjoo sub-datasets were excluded if the dataset titles indicated more targeted surveys that did not involve the potential for threatened or priority species, such as weed or pest surveys. Filtering out marine fauna was challenging as the Dandjoo fauna dataset was large (approximately 130,000 records) and lacked consistent species habitat attributes. To overcome this, habitat attributes for each species were retrieved from the World Register of Marine Species (Ahyong *et al.*, 2025) using R (R Core Team, 2025). Species classified as exclusively marine, exclusively brackish, or both were excluded from the analysis. Once the total number of records were calculated for each protected area, log-log linear models were developed (Figure A1), relating species richness to survey effort: $\log_{10}(sp.richness + 1) \sim \log_{10}(survey\ effort + 1)$.

A non-log linear model was first used, however, the variance in T&P species richness increased with higher sampling effort. As a result, protected areas with very high survey effort showed larger residuals, which could give the misleading impression that these sites were unusually T&P species-rich or species-poor. The log–log transformation was used to reduce this heteroscedasticity and produce more stable, ecologically meaningful residuals. Residuals were extracted from the model and used as effort-corrected indicators of protected area T&P species richness.

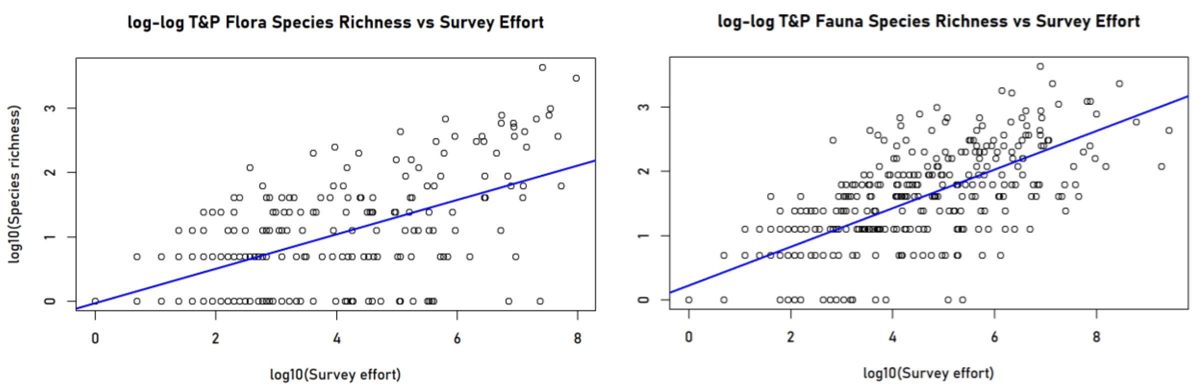

Figure A1. Log$_{10}$-log$_{10}$ linear models used to correct for sampling bias of T&P species richness. The left panel shows the model of T&P flora richness as a function of survey effort (total no. of records) within all protected areas; the right panel shows the equivalent for T&P fauna richness. Flora model: y = 0.268x – 0.0025, R^2 = 0.56. Fauna model: y = 0.301x + 0.226, R^2 = 0.63.

Wetland values

This criterion was represented by wetland management categories, which are used to provide an indication of relative wetland values in Western Australian state conservation processes (DBCA, 2025d). Conservation category wetlands (CCWs) are the highest management priority and support a high level of ecological values and functions; resource enhancement wetlands (REWs) are somewhat modified, though they retain substantial ecological attributes and functions and have the ability to be restored; and multiple use wetlands (MUWs) are significantly modified to the extent that their remaining ecological attributes and functions are limited (DBCA, 2025d).

The Geomorphic Wetlands of the Swan Coastal Plain dataset (DBCA, 2024) was used to identify wetland features within protected areas. This dataset does not extend into the Darling Scarp, so wetlands within protected areas in this region were visually identified and assigned as ‘not assessed’. Each wetland feature was assigned a unique numerical value proportional to the conservation value of each management category (5 = CCW, 4 = REW, 3 = not assessed, 2 = MUW, 1 = not applicable). Note that ‘not assessed’ wetlands were assigned a conservative, average value of three. The maximum value occurring within each protected area feature was calculated, essentially identifying the highest value wetland management category present within each protected area.

Formal protection

The ecological linkage study from O’Donnell (2020) categorised Perth’s protected areas as either ‘formal’ or ‘semi-formal’ based on their level of protection and designated land uses. The proportion of each protected area designated as formally protected was calculated using spatial datasets prepared in the above study.

Habitat area

Habitat area of each protected area was represented by the total area of intact remnant vegetation, calculated using the *Native Vegetation Extent* dataset (DPIRD, 2023). This was used as a more ecologically meaningful representation of patch size as some protected areas included sections of cleared habitat.

Longer and thinner protected areas may be less resilient than they appear due to edge effects. Accounting for 50m of edge effects was considered. However, subtracting a buffer made it difficult to determine a resilient area equivalent to 50ha, and unfairly decreased area of larger patches with wetlands which created ‘holes’ in the feature. Additionally, most long/thin protected areas are already small and thus high priority.

Regional connectivity

Hanski’s connectivity index (Hanski *et al.*, 2000) was calculated for each protected area patch as an indicator of regional connectivity. The connectivity index (*CI*) is calculated as:

$$CI_i = \sum_{j \neq i} \exp\left(-\alpha d_{ij}\right) A_j^{\beta}$$

Where d_{ij} is the shortest cumulative distance (cost-weighted) along network linkages between patch *i* and *j*, A_j is the size of patch *j* (ha), α is a decay parameter defining species dispersal ability, and β is an emigration scaling parameter defining the influence of patch size on emigration rates.

Following Rösch et al. (2013) and Gallé et al. (2022), β was set to 0.5 as this is the recommended value where analyses involve whole communities. However, their recommended α of 0.5 (based on *d* in km) resulted in too steep of a decay curve to provide meaningful variation between protected area patches in the relatively disconnected inner-urban setting – a custom α was therefore utilised (Figure A2). The α value relates species dispersal ability to the distance between patches *i* and *j* (d_{ij}), effectively communicating the probability of dispersal at distance *d*. This is represented in Hanski's calculation as $\exp(-\alpha d_{ij})$, meaning that:

$$\alpha = \frac{-\ln(P_d)}{d}$$

where P_d indicates the probability of dispersal at distance *d*. The α used by Rösch et al. (2013) and Gallé et al. (2022) corresponds to a dispersal probability of ~36.79% at 2km. In this study, α was calculated to reflect a 50% chance of dispersal at 5km (Figure A2). While this is relatively high for whole-community dispersal, it provided more meaningful variation in patch isolation at the regional scale. This provided more value as a relative indicator of patch-level connectivity for the MCDA. Instead of actual distance, cost-weighted distance (O'Donnell, 2020) was used to incorporate underlying landscape resistance and its impact on dispersal. To estimate cost-weighted distance at 5km actual distance, a $\log_{10}$-$\log_{10}$ linear model of all linkages was fitted, with cost weighted distances as the response variable (y) and absolute distances as the predictor variable (x). Logarithms were used to reduce heteroscedasticity and provide a better model fit. The cost-weighted distance was predicted for a corresponding absolute distance of 5km, giving a cost-weighted distance of approximately 75,620. With a 50% chance of dispersal, this resulted in an α of 9.166×10^{-6}.

This criterion did introduce some mild artefacts. As this network only included protected areas and not surrounding vegetation, some protected areas in highly connected surrounding landscapes appeared as isolated from others, where at a landscape scale they were well-connected. Protected area patches were also represented as nodes without spatial extent. When calculating cumulative distances between patches, this tended to overestimate connectivity in patches near large or elongated ones, since internal distances were excluded from the calculations. Lastly, protected areas near the edge of the study area had slightly lower connectivity values as patches outside of the study area were not included.

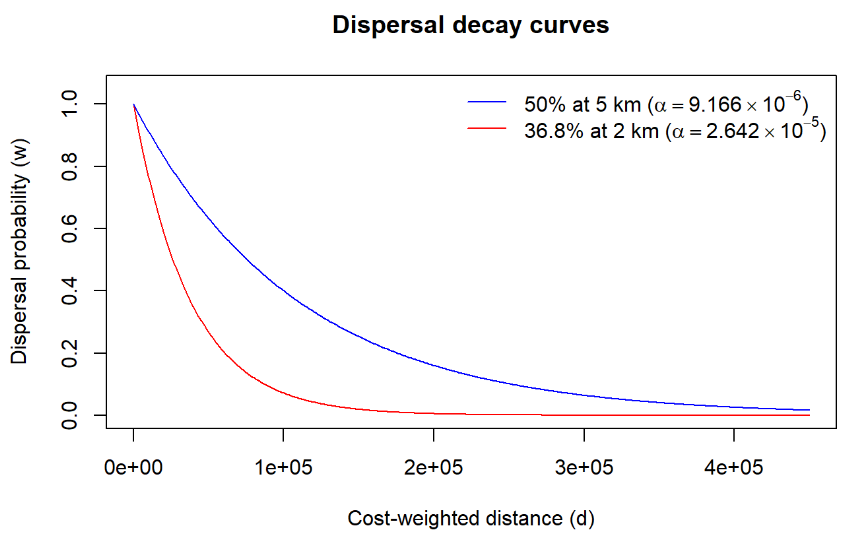

Figure A2. Dispersal decay curves across the range of cost-weighted distances present in the Perth-Peel ecological network. Each line represents the result of a unique alpha value; the blue line shows the alpha value used in this analysis, while the red line represents the alpha value corresponding to Gallé et al. (2022), adjusted to cost-weighted distance.

Linkage length

Linkage lengths (m) were extracted directly from the spatial datasets prepared by O'Donnell (2020).

Permeability

The average cost per metre of each linkage was derived from O'Donnell's (2020) spatial datasets, by dividing the cost-weighted distance by the total length.

The cost layer itself was subject to data limitations, and included some simplifications as a result. However, this was unlikely to have significantly affected broad average cost values and so the cost layer was used without alteration.

Maximum gap in habitat

The maximum distance between areas of remnant vegetation was calculated for each linkage. This was achieved by splitting the linkages (O'Donnell, 2020) as they intersected patches of remnant vegetation (DPIRD, 2023), calculating the length of each linkage segment, and selecting the maximum value for each linkage.

Barriers crossed

Barriers were defined using the ecological linkage layer and cost raster layer from O'Donnell (2020). Following this study, barriers were defined as any pixel ≥100 in the cost raster. This included open water, state roads, freeways, rail corridors, and airports. Cost profiles were calculated for each linkage feature in QGIS using the Profile Tool 4.2.6 plugin and imported to Microsoft Excel to calculate the total number of unique barriers for each linkage. Unique barriers were defined as pixels along the profile that were (a) a barrier, (b) not immediately following a barrier pixel, and (c) not the first pixel in a new linkage. This excluded the over-counting of barriers which were multiple pixels wide.

Some barriers may have been double counted as a result if they were in close spatial proximity (e.g. divided state roads), while one very wide barrier will be counted as one (e.g. freeway). A buffer was considered to prevent small barrier gaps from double counting. However, this also removed areas where distinct but closely spaced barriers genuinely occurred. Given this only occurred in a few edge cases, this limitation was accepted.

Surrounding vegetation

The *rehabilitation suitability index* for surrounding vegetation was calculated using a raster layer representing the rehabilitation suitability of four different vegetation classes. Suitability scores were given based on the relative need for and feasibility of ecological rehabilitation (i.e. enhancing habitat quality) within each type of vegetation (Table A1). Remnant vegetation represents areas which already contain habitat value (although sometimes degraded) but are relatively easy to rehabilitate; perennial and annual vegetation represent areas with higher need for habitat but still a relatively high feasibility of rehabilitation, giving them a moderate to high suitability score; and no vegetation represents areas with low habitat value but often consisting of buildings or asphalt, making them often difficult to rehabilitate. Perennial and remnant vegetation were both given a feasibility score of 1 given they have existing vegetation providing a baseline level of habitat.

Table A1: Rehabilitation suitability scores of four different vegetation classes. Each vegetation class was given a 'need' and 'feasibility' score between 0 and 1, from the perspective of ecological rehabilitation. The suitability score was calculated as $need\ score \times feasibility\ score$.

Class	Need score	Feasibility score	Suitability score
Remnant vegetation	0.2	1	0.2
Perennial vegetation (non-remnant)	0.6	1	0.6
Annual vegetation (non-remnant)	1	0.8	0.8
No vegetation	1	0.3	0.3

A raster layer of these vegetation classes was created using four raster inputs at 10m spatial resolution. Normalised difference vegetation index (NDVI) was used index to identify and classify the vegetation types, given its wide applicability as a measure of vegetation greenness (Huang *et al.*, 2021). $NDVI_{winter}$ was calculated using Sentinel 2B imagery from the 6th August 2024, while $NDVI_{summer}$ was calculated using Sentinel 2B imagery from the 1st March 2025, before the first rains (ESA, 2025). Cells which exhibited greenness year-round were classified as perennial vegetation, while those that were green during winter but not summer were classified as annual vegetation. The *lawn* layer was used to identify annual vegetation which would be classified as perennial by NDVI due to it being watered through summer and retaining its greenness. *Lawns* were represented using a random-forest-classified binary raster sourced from Peterson and Andrew (2025); and r*emnant vegetation* was represented by a binary rasterisation of the *Native Vegetation Extent* dataset (DPIRD, 2023). Each vegetation class was defined as:

- Remnant vegetation = *remnant vegetation*

- Perennial vegetation = $NDVI_{winter}$ ≥ 0.2 AND $NDVI_{summer}$ ≥ 0.2 AND is not *remnant vegetation* AND is not *lawn*
- Annual vegetation =($NDVI_{winter}$ ≥ 0.2 AND $NDVI_{summer}$ ≤ 0.2) OR is *lawn*
- No vegetation = is not *remnant vegetation* AND is not *perennial vegetation* AND is not *annual vegetation*

Each pixel in the rehabilitation suitability raster was assigned a value equal to the suitability score of its vegetation class. The *rehabilitation suitability index* was then specified for each linkage by calculating the mean of all pixels within a 100m buffer of the linkage.

APPENDIX B – WILDLIFE GUILD LITERATURE OVERVIEW

Guild	Representative species	Habitat factors	Movement factors
Tree birds	• Carnaby's Black Cockatoo (*Calyptorhynchus latirostris*)* • Red-tailed Black Cockatoo (*Calyptorhynchus banksii naso*)* • Grey Butcherbird (*Cracticus torquatus*) • Tawny Frogmouth (*Podargus strigoides*) • Tree Martin (*Hirundo nigricans*)	• A study in the Perth region classified a range of species as tree-dependent – including galahs and the grey butcherbird. The study found species richness was positively related to amount, type and condition of native vegetation (Davis *et al.*, 2013). The group generally required trees for feeding, nesting or perching. Species feed in the canopy or were hollow nesters that feed on the ground (galah) (Davis *et al.*, 2013) • Carnaby's, forest red-tailed, and Baudins black cockatoos are all heavily associated with large eucalypts for nesting, breeding and foraging (Chapman, 2008; Saunders *et al.*, 2014). • Black cockatoos utilise vegetation dominated by Banksia spp., Tuart woodlands, as well as Marri and Jarrah forests (DAWE, 2022). They also use tall trees in and around riparian environments of permanent water sources to roost (DAWE, 2022). • Grey butcherbird utilises eucalypt forests and woodland with sparse shrub layer and dense to no understorey layer. Often found in or at edges of remnant vegetation (Marchant *et al.*, 1990). • The tawny frogmouth requires tall trees, preferring eucalypt woodlands or forests with areas of open ground. Can be found in garden and street trees in suburbs (Marchant *et al.*, 1990), suggesting they are utilise the urban environment.	• Movement ecology literature was heavily concentrated on black cockatoos. • Environmental Protection Authority guidance states Carnaby's cockatoo require food and water within 6km (EPA, 2019). This is a suitable maximum distance to maintain habitat connectivity throughout the landscape. • Carnaby's cockatoo flocks follow vegetation corridors and actively avoid cleared and open areas (EPA, 2019). • Juvenile galahs in WA wheatbelt have been found to disperse widely. One study found most flew less than 20km before death, with 1/3 flying less than 5km from the capture point (Rowley, 1983). • Black cockatoos can forage up to 12km away from nesting trees during breeding seasons, up to 20km from roosting habitat during non-breeding season, and roost sites are usually within 2km of a watering point (DAWE, 2022). • Forest red-tailed black cockatoos have been found to move between 4-16.5km per day (Shephard and Warren, 2019; Rycken *et al.*, 2022). However they do so slowly, with many small relocations of an average of 48m apart (taking an average of 13min each) during most of the day (Shephard and Warren, 2019). • In the Perth region and surrounds, black cockatoo roosting sites mainly occurred in public green space and private property closely spatially associated with foraging habitat. Riparian zones and roadside vegetation were crucial for foraging and as connective landscape structures (Rycken, 2019; Rycken *et al.*, 2022) • Black cockatoo movement is driven by food resources in the landscape. Dispersal seems to be variable depending on how limited food is, with less food resulting in further travel in search for it (Rycken, 2019).

Woodland birds	• Inland thornbill (*Acanthiza apicalis*) • Rufous Whistler (*Pachycephala rufiventris*) • Splendid Fairy-wren (*Malurus splendens*) • Western Spinebill (*Acanthorhynchus superciliosus*) • Western Thornbill (*Acanthiza inornata)* • Western Wattlebird (*Anthochaera lunulata*) • White-browed Scrubwren (*Sericornis frontalis*) • Western Wattlebird, Djoongong (*Anthochaera lunulata*)	• A study in the Perth region classified a range of species as 'sensitive to vegetation cover', including a range of small bushland birds. Species richness was positively related to the amount, type and condition of native vegetation (Davis *et al.*, 2013). Richness was also positively related to proportion of native vegetation, proportion of other vegetation, tree and canopy cover, hollow presence, and wetlands. However, these are all very interrelated. The most parsimonious model found that the proportion native and other vegetation within 2km accounted for 57% of the total deviance in species richness (Davis *et al.*, 2013) • All of these species are uncommon in urban sites, appearing more in vegetated landscapes (Davis *et al.*, 2013). However, many are known to utilise non-native or isolated trees and shrubs in golf courses or gardens, including the western spinebill and white-browed scrubwren (Marchant *et al.*, 1990). • Variety of the species require or prefer dense understorey in forest/woodland, including the splendid fairywren, white-browed scrubwren, inland thornbill, and western spinebill, while others are associated with forest/woodland habitats with diverse understoreys (e.g. western wattlebird) (Marchant *et al.*, 1990). However, all are to some extent dependent on understorey vegetation. • Species are also often associated with wetlands and waterways (Marchant *et al.*, 1990)	• (Watson *et al.*, 2001) rural area study. Thresholds for isolation for woodland birds: Grey fantail: 400m Superb fairy wren: 400m Yellow rumped thornbill (and other thornbill species): 400-500m White-browed scrubwren: 300m Rufous whistler: 300m • Likely to be obstructed by smaller buildings and roads due to the tendency for these species to inhabit lower vegetation strata and altitudes (Marchant *et al.*, 1990). • While many passerines such as wrens and other bushland birds have a median dispersal distance of up to 10 home ranges from their birthplace, cooperative breeders such as Australian wrens are highly philopatric and tend to be between 2 and 4 home range widths (Russell and Rowley, 1993). For urban superb fairy wrens with home ranges of 1.4 ha (Parsons, 2024), this relates to a dispersal distance of approximately 170 - 530m. • One study in rural WA found blue-breasted fairy wrens could travel multiple kilometres between patches, however out of 302 birds over three years, only seven made these interpatch dispersals (Brooker and Brooker, 1997).
Terrestrial fauna	• Bobtail (*Tiliqua rugosa*) • Perth Slider, Lined Skink (*Lerista lineata*)* • Quenda (*Isoodon fuscivente*r)* • Soft spiny-tailed gecko (*Strophurus spinigerus*) • Western brush wallaby (*Notamacropus irma*)* • Honey possum (*Tarsipes rostratus*) • Bush rat (*Rattus fuscipes*)	• Quendas inhabit dense understorey in banksia or jarrah woodlands, or around swamps. Nesting sites are indentations in the ground and they are generally hidden beneath shrubs and lined with soft vegetative material (DBCA, 2017a). • Quendas also utilise non-native vegetation and properties with shrubs and suitable habitat for nesting and foraging (DBCA, 2017a). • Study on quenda on the Murdoch University campus found that quenda slightly preferred areas with lower grass and litter/canopy cover, strongly preferring dense shrub cover (Watson, 2018) • The western brush wallaby commonly inhabits dry sclerophyll forest, and banksia woodlands and shrublands. They prefer dense, low vegetation providing adequate cover (Everard and Bamford, 2016).	• *Isoodon obesulus obesulus* move 350m up to 700m (males) in rural settings (SA and SE Aus) (Paull, 1995; Robinson *et al.*, 2018). • *Isoodon fuscivente*r travelled an average of 90m within an urban-situated site in Perth, with males on average travelling further than females. Maximum long-range distance travelled was 950m by a male, with other long range distances included 274m (female with pouch), 307m (male) and 502m (male) (Clunies-Ross and Clark, 2011). • Quenda mean maximum dispersal distance was 188m (male) and 109m (female) in a rural reintroduction study in a West-Australian wheatbelt remnant (Short, 2024). • Western brush wallabies have large home ranges. For males, this is up to 9.9ha in a night (approximately 350m diameter) and 69.2ha in a year, while for females it is up to 5.3ha in a night and 32.5ha in a year (Everard and Bamford, 2016), though this is within habitat.

		• Bobtails shelter in dead vegetation, under rubbish and in burrows. They are common in pockets of bushland and utilise urban suburbs (Bush *et al.*, 2010). • Bobtails can also utilise low-cover areas as long as they provide protection from temperature variation and predators (Millsteed, 2025). • Bush rats live in dense understorey and prefer non-settled areas (Nevill *et al.*, 2014). • Narrow strips of vegetation (especially degraded or with insufficient cover) may act as population sinks (ecological traps) for quendas and movement corridors for foxes (a main predator) (Howard *et al.*, 2014), highlighting the need for dense vegetation and adequate vegetation condition.	• Honey possum dispersal is generally low <30m in Fitzgerald River National Park (Garavanta *et al.*, 2000), although a smaller and later radio-telemetry study in same area found that some males may move much further, potentially up to 200m (Bradshaw and Bradshaw, 2002). • Individual reptile movements within Bold Park, Perth, as part of a multi-decade capture-recapture study: *Ctenotus fallens* (long-tailed ctenotus) 1126 ± 567 m *Strophurus spinigerus* (southwest spiny tailed gecko) 1164 ± 420 m *Pogona minor* (western bearded dragon) 1207 ± 458 m *Cryptoblepharus buchananii* (snake eyed skink) 400 ± 0 m *Tiliqua rugosa* 1414 ± 308 m *Ctenotus australis* (Western limestone ctenotus) 1615 ± 50 m *Ctenophorus chapmani* (heath dragon) 1180 ± 430 m *Cyclodomorphus celatus* (W slender bluetongue) 750 m (How, 2024) • A non-published study investigated the frequency of bobtail sightings at different distances from bushlands in Perth's urban areas. They found "frequencies of 21, 37, 86, and 59 Bobtails for distances of up to 50m, 100m, 500, and 1km respectively… A total of 59 Bobtails were seen beyond the 1km mark" (Falla, 2022) • Within habitat, bush rat territory is within a 200-250m radius. Males extend their range in spring, travelling up to 1km in a night (Nevill *et al.*, 2014). • Causes of quenda mortality in a 2012 citizen-science study (Howard *et al.*, 2014): 44% vehicle strike. 30% predation (98% from cats, foxes and dogs). 16% drowning (pools, fish ponds or drainpipes). Bobtail mortality (Millsteed, 2025): 30-40% Penetrating trauma, mostly from dogs. 16-24% blunt trauma, cars or garden equipment.

Wetland fauna	• Australasian bittern (*Botaurus poiciloptilus*)* • Clicking froglet (*Crinia glauerti*) • Moaning frog (*Heleioporus eyrei*) • Snake-necked turtle (*Chelodina oblonga*) • Water rat, Rakali (*Hydromys chrysogaster*)* • Western swamp tortoise (*Pseudemydura umbrina*)*	• Southwest snake-necked turtle associated with permanent wetlands, and less commonly with seasonal wetland. Females lay eggs on land (Nevill *et al.*, 2014). • Snake-necked turtles also strongly associated with fringing vegetation. While nesting in relatively bare ground, canopy cover provides ideal microclimate regulation while understorey cover provides protection from predation while travelling to and from nesting sites (Santoro *et al.*, 2020, 2024). • Water rats nest in small tunnels along watercourses and lakes, and feed on aquatic insects, crustaceans and fish, and sometimes frogs, lizards or small mammals or birds may be eaten (Nevill *et al.*, 2014). • Perth's frogs often require shade, damp microclimates and shelter provided by dense understorey, logs and litter (Bush *et al.*, 2010). • Residential gardens are commonly used by snake neck turtles as nesting sites and positively associated with species abundance, though this is likely not due to direct association but rather an association with historical land use (native vegetation) (Santoro *et al.*, 2020).	• Southwest snake-necked turtles have been generally observed between 0-200m, but up to 500m, from the water's edge of wetland habitats (Bartholomaeus, 2016). • Frog dispersal is not well studied. A distance of 1000m was used by Kirk et al. (2018) as a dispersal threshold in a connectivity study in Melbourne. In urban Melbourne, the growling grass frog disperses no more than 500m from tagged sites (Heard *et al.*, 2012), while other species can travel multiple kms (Morris, 2012). • Water rat movements have been recorded up to 3.1km in less than 6 hours (Gardner and Serena, 1995), and over 3 km overnight (Vernes, 1998), however these are generally along the water's edge of rivers and within habitat. • A Perth study found snake-necked turtles nest an average of 69m from the water, with a range of 2m-703m (Santoro *et al.*, 2024). • Western swamp tortoises can travel up to 600m over 2 days, 450m over 1 day, and animals found up to 1km outside nature reserve boundaries (likely during search for optimal habitat) (Burbidge *et al.*, 2010). However, this species is very uncommon in Perth.

*species of conservation significance

APPENDIX C – LANDSCAPE FEATURE DATASETS AND PROCESSING

Focal layer inputs				
Landscape feature	**Description**	**Datasets**	**Processing details**	**Notes**
Habitat barriers	Roads and railway lines that are likely to separate patches of habitat	• Road Network (Main Roads, 2025) OpenStreetMap rail data: July 2025 (OpenStreetMap Contributors, 2025)	State roads were buffered by 10m, while local roads and rail lines were buffered by 7m. All layers were merged and dissolved.	OpenStreetMap data screened for quality purposes and was used for its higher accuracy than other available data.
Native vegetation patches	Remnant native vegetation polygons that have undergone digital smoothing to represent contiguous focal patches.	• Native vegetation extent (DPIRD, 2023) • *Habitat barriers* layer	Remnant patches were first dissolved to merge contiguous vegetation. Patches separated only by tracks or thin corridors (<20m wide) were joined by applying a 10m and then -10m buffer. Any patches that connected across roads or railway lines during this process were split again by masking the vegetation patches with the habitat barriers layer. To exclude thin patches subject to intense edge effects and likely of degraded condition, a -15m buffer followed by 15m buffer was applied, removing areas less than 30m in width.	-
Black cockatoo nesting and roosting sites	Confirmed nesting or roosting locations of Carnaby's or forest red-tailed black cockatoos	• Threatened species and communities database (DBCA, 2025c)	Confirmed black cockatoo nesting and roosting points were filtered for ones that had been used (maximum bird count >0). Canopy height model raster layers of trees (>3m) were converted to vector points. All tree points within 50m of a black cockatoo nesting or roosting point were selected and buffered by 20m to join separate trees into one polygon.	-
Conservation category wetlands	Wetlands designated in the 'conservation' management category by DBCA under state policy	• Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024)	-	-
Protected areas (NatureLink)	Protected areas as defined by O'Donnell (2020)	• NatureLink Perth protected areas data (O'Donnell, 2020)	-	-

Resistance layer inputs (all layers converted to binary presence/absence raster at 10m resolution)				
Landscape feature	**Description**	**Datasets**	**Processing details**	**Notes**
Land cover				
Remnant vegetation	Native remnant vegetation or natural regrowth	• Native vegetation extent (DPIRD, 2023)	-	This layer was used to differentiate planted and remnant vegetation in the vegetation features below, which informed relative resistance values.
Tall trees	Presence of tall trees (>8m)	• Canopy height model (CHM) Perennial groundcover: • Sentinel 2B satellite imagery: dates 6/08/2024 and 1/03/2025 (ESA, 2025) • Random forest-classified lawn data sourced from Peterson and Andrew (2025)	CHM was aligned and resampled from 1m to 10m resolution using an average. Converted to binary raster where presence was defined as cover of 20% or more in a cell.	-
Trees	Presence of trees (3-8m)		CHM was aligned and resampled from 1m to 10m resolution using an average. Converted to binary raster where presence was defined as cover of 20% or more in a cell.	-
Shrubs	Presence of shrubs (1.5-3m)		CHM was aligned and resampled from 1m to 10m resolution using an average. Converted to binary raster where presence was defined as cover of 10% or more in a cell.	Low value of 10% was chosen to enable presence of midstorey and understorey layers to be detected where canopy masks some of the cell.
Subshrubs	Presence of subshrubs (0.5-1.5m)		CHM was aligned and resampled from 1m to 10m resolution using an average. Converted to binary raster where presence was defined as cover of 10% or more in a cell.	Low value of 10% was chosen to enable presence of midstorey and understorey layers to be detected where canopy masks some of the cell.
Groundcover	Presence of perennial groundcover (0-0.5m)		CHM was aligned and resampled from 1m to 10m resolution using an average. Converted to binary raster where presence was defined as cover of 10% or more in a cell. Filtered for perennial (as opposed to annual) groundcover using NDVI - cells must be green (NDVI>0.2) in both winter and summer, and cannot be lawns.	Low value of 10% was chosen to enable presence of midstorey and understorey layers to be detected where canopy masks some of the cell.

'Other' perennial vegetation	Perennial vegetation, including deciduous trees, derived from NDVI data. This accounts for perennial vegetation not identified by the canopy height model, albeit without height data	• Sentinel 2B satellite imagery: dates 6/08/2024 (winter) and 1/03/2025 (dry – just after summer before rains) (ESA, 2025) • Random forest-classified lawn data sourced from Peterson and Andrew (2025) • Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024)	Defined as cells that are: • Green (NDVI>0.2) in both winter and summer, OR • Deciduous (green in summer but not winter, given they are not within a perennial or seasonal wetland feature - see 'annual vegetation' below), AND • Not lawns	-
Annual vegetation	Annual vegetation (grasses, herbs), including summer-green wetland vegetation		Defined as cells that are: • Green (NDVI>0.2) in winter but not summer, OR • Lawns, OR • Wetland summer-green vegetation (green in summer but not winter, given it is within a perennial or seasonal wetland feature)	-
Bare ground	Non-vegetated areas (bare soil, asphalt, roofs)		Defined as cells that are: • Not perennial vegetation, AND • Not annual vegetation	-
Open water*	Exposed surface water from perennial water features	• Water Observations from Space (Geoscience Australia, 2024)	-	Water Observations from Space dataset was taken from a summer date to approximate perennial water sources.
Wetland water	Open water within perennial or seasonal wetlands with a minimum of moderate conservation value ('resource enhancement' management category wetlands assigned by DBCA under state policy)	• Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024) • Water Observations from Space (Geoscience Australia, 2024)	Filtered for conservation and resource enhancement category wetlands. Perennial and seasonal wetlands were defined using the official wetland classification types defined by state government (DBCA, 2025d). Selected features were masked by the *open water* layer.	-

Land use				
Additional protected areas	Native vegetation and wetlands that have a level of protection	• Geomorphic Wetlands of the Swan Coastal Plain (DBCA, 2024) • Native vegetation extent (DPIRD, 2023) • Bush Forever (DPLH, 2019) • Regional Parks (DBCA, 2023) • Ramsar Sites (DBCA, 2017b) • Legislated Lands and Waters (DBCA, 2025b)	Defined as: Conservation category wetlands Any regional park Bush Forever areas (native vegetation only, to remove areas that have been cleared/developed since 2000) Any Ramsar site Legislated lands and waters on Crown land (i.e. excluding freehold)	-
Drainage channels	Open drainage channels	• Drainage Open Channel (Water Corporation, 2025)	Rasterised using ALL_TOUCHED option**	-
Airports	Land areas and infrastructure used for aviation	• OpenStreetMap aeroway data: July 2025 (OpenStreetMap Contributors, 2025)	Excluded helicopter landing pads	OpenStreetMap data screened for quality purposes and was used due to limitations in accessing airport data.
Parks and forestry	Public green space and managed forestry areas	• Urban forest mesh blocks (DPLH, 2025)	-	-
Rural	Primary production land and agricultural areas		-	-
Street block	Residential or commercial areas		-	-
Campus	Institutional grounds for education, hospitals and some recreation		-	-
Industrial	Land zoned for industrial operations		-	-
Other	Some transport and utility areas, including road corridors		-	-
Barriers				

Transmission overhead powerlines	Large, high-voltage transmission powerlines supported by towers	• Transmission Overhead Lines (Western Power, 2025)	Rasterised using ALL_TOUCHED option**	-
Rail corridors	Land reserved for rail and train operations	• OpenStreetMap rail data: July 2025 (OpenStreetMap Contributors, 2025)	Rasterised using ALL_TOUCHED option**	OpenStreetMap data screened for quality purposes and was used for its comprehensiveness over official data sources (local transit authority data only included publicly owned and managed rail corridors).
State roads	Major state government-owned roads designed for large volumes of traffic	• Road Network (Main Roads, 2025)	Filtered for state roads. Rasterised using ALL_TOUCHED option**	-
Local roads	Local government-owned roads designed for local traffic at low to moderate volumes		Filtered for local roads. Also includes crossovers and miscellaneous roads. Rasterised using ALL_TOUCHED option**	-
Culverts	Under-road tunnels or channels designed for water drainage	• Culverts (Main Roads, 2024) Road Network (Main Roads, 2025)	Culvert point data buffered by 10m and rasterised. This was overlaid with local/state roads (see above) to select only road cells with culverts underneath.	-
Buildings	Permanent human-made structures with walls and a roof	• Buildings of WA (DPIRD, 2022)	-	Machine learning identification of buildings was not completely accurate, however for the use case it approximated building locations and density well.

*Landscape feature not always within this category, depending on the guild.

**Most vector layers were rasterised using the GDAL vector conversion tool with standard settings (burn value where feature crosses centroid). Some layers utilised the ALL_TOUCHED option which burns a value for every cell touched by a vector line. This ensured no diagonal 'gaps' in the resulting raster file.

APPENDIX D – DIGITISATION AND ATTRIBUTION PROCEDURES

Digitisation conventions

General

- Utilise high resolution historical aerial imagery from Landgate, Google Street View imagery, and canopy height model data to inform digitisation.
- Digitise at a scale of approximately 1:5000. Utilise scales between 1:2500 – 1:10000 to maintain a balance between finer detail and larger landscape context where necessary.
- Draw feature boundaries along clear delineations in the landscape. This may include along barriers or tracks, the edge of relatively continuous canopy, or other cues which clearly signify the boundary between a focal patch and non-focal areas.
- Provide an uncertainty score on each manually attributed or digitised feature (Table D1) and provide relevant notes.

Table D1. Confidence scoring system applied to manually digitised or attributed features, showing score values, their meaning, and descriptions.

Score	Meaning	Description
0	Confident	General vegetation type, condition and structure clearly identifiable from desktop sources, with clear and recent satellite and street view imagery of the area.
1	Low	Some minor ambiguities in general vegetation type, condition or structure in small parts of the patch due to somewhat poor or outdated imagery, though attributes are mostly discernible.
2	High	General vegetation type, condition or structure cannot be reliably attributed from desktop assessment due a lack of clear and/or current imagery for large parts of the patch, or the presence of conflicting data.

Focal patches

- Extend a patch boundary where the mapped edge clearly misses contiguous habitat >0.5ha.

- Trim a patch boundary where the mapped edge clearly extends beyond contiguous habitat >0.5ha.
- Join two nearby (<100m) patches where mapped patch boundaries do not meet but are separated only by small breaks in contiguous habitat (<30m), given no major barriers separate the patches. This prevents excessive current flow and loss in the connectivity assessment between two patches that are already essentially connected.
- Digitise a new focal patch where there is no mapped feature, but evidence suggests the area meets the ecological criteria for a given guild.

Vegetation strata

- Draw features around significant areas of discrepancy (>0.5ha) within the canopy height model (CHM) data.
- Take care to follow clear landscape cues and stay within the area of error in the CHM data.

Vector attribution templates

Focal patches

Field	**Description**	**Options**
site	Name of the local study area	*Bibra-Piney* *Herron Pt* *Piara*
date	Latest date of digitisation and attribution	
focal	Whether the feature meets the criteria to be deemed a focal patch	*Y* *N*
veg_type	Dominant broad vegetation type	*Banksia woodland* *Eucalypt woodland/forest* *Melaleuca woodland* *Shrubland* *Coastal heath*
veg_cond	Dominant vegetation condition, using the vegetation condition scale adapted from Keighery (1994). Scale is simplified to two categories for ease of assessment: D: degraded (or completely degraded) G: good or better	*D* *G*

dt_uncert	Uncertainty level related to patch boundary or attribution based on the desktop assessment.	*0* *1* *2*
evidence	Source of the evidence used for the new patch boundary and attribution.	*Desktop* *Field*
notes	Any relevant notes related to the patch shape, attribution, source(s) of evidence, or uncertainty.	

Vegetation strata

Field	**Description**	**Options**
site	Name of the local study area	*Bibra-Piney* *Herron Pt* *Piara*
date	Latest date of digitisation and attribution	N/A
canopy	Confirmed presence/absence of canopy layer (>3m) in the area *0*: Layer absent – overlay CHM data *1:* Layer present (>20% cover) – overlay CHM data *NULL:* Layer sparse (>20% cover) – do not overlay CHM data	*0* *1*
shrub	Confirmed presence/absence of shrub layer (1.5-3m) in the area *0*: Layer absent – overlay CHM data *1:* Layer present (>20% cover) – overlay CHM data *NULL:* Layer sparse (>20% cover) – do not overlay CHM data	*0* *1*
subshrub	Confirmed presence/absence of subshrub layer (0.5-1.5m) in the area *0*: Layer absent – overlay CHM data *1:* Layer present (>20% cover) – overlay CHM data *NULL:* Layer sparse (>20% cover) – do not overlay CHM data	*0* *1*
gcover	Confirmed presence/absence of groundcover layer (0-0.5m) in the area *0*: Layer absent – overlay CHM data *1:* Layer present (>20% cover) – overlay CHM data *NULL:* Layer sparse (>20% cover) – do not overlay CHM data	*0* *1*
dt_uncert	Uncertainty level related to feature boundary or attribution based on the desktop assessment.	*0* *1* *2*
evidence	Source of the evidence used for the feature boundary and attribution.	*Desktop* *Field*
notes	Any relevant notes related to the feature boundary, attribution, source(s) of evidence, or uncertainty.	N/A