Restoring Longitudinal Connectivity of an Anthropogenically Fragmented Stream Network: The Hunter River Catchment

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ABSTRACT

Connectivity is a fundamental attribute of riverine landscapes, critical to the functioning of freshwater ecosystems. Riverine landscapes are longitudinally connected and at the drainage basin scale the network of streams provide pathways for the downstream dispersal of water, sediments, organisms and nutrients, while also connecting freshwater habitats. A multitude of anthropogenic barriers, including dams, weirs, floodgates and road crossings have disrupted connections and natural processes within stream networks. Fragmentation has compromised the integrity of, and connections within, freshwater ecosystems worldwide. Improving longitudinal connectivity through activities such as barrier removal has become a common objective of river ecosystem restoration. However, river restoration projects are commonly implemented over relatively small scales. The need for restorations to be carried out within a catchment context is increasing in prominence but the knowledge and approaches to restoring connectivity of entire stream networks remains limited.

This thesis contributes to our understanding of restoring riverine connectivity within the context of drainage basins. It presents an optimisation-based approach to stream network connectivity restoration that is robust and applicable to rivers and ecosystems worldwide. This is in contrast to the majority of current approaches that are focused at-a-site and on northern hemisphere fish species. The findings of this thesis have significance to river science and management, by identifying the important regions for restoring connectivity in the Hunter River catchment. In addition, it elucidates the influence of riverine landscape character on optimally restoring stream network connectivity.

The research presented focuses on restoring the longitudinal connectivity of the anthropogenically fragmented Hunter River network in New South Wales. A generic optimisation model was developed to identify the optimum combination of barriers to be removed or altered so as to achieve maximum improvements in connectivity of an entire stream network within a given restoration budget. Stream network connectivity was quantified with the habitat-quality modified Dendritic Connectivity Index for both diadromous and potamodromous fish species. A desktop-based approach was employed, where first, a GIS approach was used to identify all in-stream anthropogenic barriers in the network. Second, the output sensitivity of the optimisation model to stream network and barrier character was assessed on the Williams River, a sub-catchment of the Hunter River catchment. Third, this model was applied to identify those barriers and regions of the stream network important to restore connectivity within the entire Hunter River network.

Longitudinal connectivity of the Hunter River network has been fragmented by a potential 2466 anthropogenic barriers of different barrier types, substantially dominated by road crossings. Barriers located on the majority of the main waterways, including the Hunter and Goulburn Rivers, and in some of the sparsely-fragmented western streams were critical to restoring connectivity of the Hunter River network. These barriers were predominantly composed of road crossings and a few weirs were also regularly selected. Removing five percent of optimally selected barriers could reconnect up to 23 percent of the stream network.

Optimally improving longitudinal connectivity of stream networks was intricately linked to which barriers were selected. Generally, the largest improvements in stream network connectivity occurred following modification of barriers selected earliest in the optimisation procedure, after which the rate of improvement steadily decreased with further barrier selections. In addition, restoration of connectivity was more rapid for diadromous than potamodromous fish species. Restoring stream network connectivity was dependent upon and varied according to the character of the stream network and in-stream barriers, with barrier character having a substantial influence on connectivity while habitat quality was found to be insignificant. Despite this, the spatial organisation of the stream network and its barriers were central to initial barrier selection, irrespective of the character of the Williams River system. Barriers located on the main channel in the downstream reaches close to the river mouth were critical to restoring connectivity for diadromous species, while barriers located on the main channel but in the central regions of the river network were important for potamodromous migrations.

CERTIFICATE OF AUTHORSHIP OF THESIS

I certify that the substance of this thesis has not already been submitted for any degree and is not currently being submitted for any other degree or qualification.

I certify that any help received in preparing this thesis and all sources used have been acknowledged in this thesis.

Signature of Author: [redacted]

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Chapter 1

1 Chapter 1 – Introduction

1.1 BACKGROUND AND CONTEXT

Connectivity is a fundamental attribute of riverine landscapes (Wiens, 2002), with water mediating connections between different components of the system. Hydrological connectivity provides pathways for the transfer of matter, energy and organisms among different components of the river system (Pringle, 2001), which is critical to the integrity of freshwater ecosystems (Pringle, 2003). The riverine landscape is hydrologically connected in three spatial dimensions, longitudinal, lateral and vertical, with the nature and extent of connections varying over time, representing the fourth dimension of connectivity (Ward, 1989). The spatial configuration and connectivity between riverine habitats is important, as they influence hydrological, geomorphological and ecological processes (Power and Dietrich, 2002; Benda *et al.*, 2004; Campbell Grant *et al.*, 2007; Rodriguez-Iturbe *et al.*, 2009). For example, the branching spatial configuration and connectivity of stream networks provides migratory pathways and substantially confounds movement and dispersal of water-bound species (Fagan, 2002; Ganio *et al.*, 2005; Campbell Grant *et al.*, 2007), especially species that undergo obligatory large scale migrations for life cycle completion, such as migratory fish. In addition to spatial connections in the riverine landscape, the temporal variability and timing of these spatial connections is also ecologically significant (Poff *et al.*, 1997). The variable nature of hydrological connectivity shapes the riverine landscape, increases landscape heterogeneity, provides natural cues to fauna and flora within riverine ecosystems and, the dynamic nature of connectivity is critical to the functioning of wetland ecosystems (Junk *et al.*, 1989; Amoros and Bornette, 2002).

In an attempt to control flow and secure critical water resources, humans have modified the character of hydrological connections in riverine landscapes worldwide. More than 45 000 large dams have been constructed internationally (World Commission on Dams, 2000), with over half of the large drainage basins in the world impacted by flow regulation as a result of dams (Nilsson *et al.*, 2005). Dams influence freshwater ecosystems by interrupting downstream processes and through barrier effects. Impoundment results in three orders of impact on rivers (Petts, 1984). First-order impacts alter the downstream transfer of water and sediments (Ward and Stanford, 1983; Ward and Stanford, 1995). These alterations in downstream connectivity result in second-order impacts, in the form of changes to the physical structure of riverine landscapes. Third-order impacts reflect the changes to freshwater ecosystems as a consequence of the first two orders of impact. In addition, these structures act as permanent barriers to within-channel species migrations. Longitudinal connectivity can also be temporally or partially restricted by smaller, but substantially more abundant, structures; including road crossings (Warren and Pardew, 1998; Foster and Keller, 2011), floodgates (Kroon and Ansell, 2006) and smaller water retaining structures such as milldams and weirs (O'Connor *et al.*, 2006; Walter and Merritts, 2008). Globally freshwater ecosystems are commonly impacted by anthropogenically induced changes to connectivity (Pringle *et al.*, 2000b; Bunn and Arthington, 2002).

Connectivity of freshwater habitats have been fragmented and reduced by a plethora of anthropogenic barriers worldwide (Nilsson *et al.*, 2005) and in Australia (Harris, 1984). Fragmentation is synonymous with a landscape scale process resulting in habitat change (Forman, 1995; McIntyre and Hobbs, 1999) and associated with decreased habitat connectivity (Fahrig, 2003). The streams of the mid-Atlantic, USA, have been fragmented by more than 65 000 milldams, with the average spacing between these barriers being as low as 2.4 km (Walter and Merritts, 2008). Australian river systems have also been fragmented by thousands of water retaining structures since European settlement, which were built to secure highly variable water resources for human consumption (Arthington and Pusey, 2003). In the coastal catchments of New South Wales, Australia, Harris (1984) estimated that 11 120 km or 32 percent of stream lengths are fragmented by 293 anthropogenic barriers. In addition, these catchments have been further fragmented by numerous other smaller non-permanent barriers, often constructed on waterways for various purposes such as providing safe passage across channels or protection from tidal surges. Williams and Watford (1997) identified 185 culverts, five fords, 46 causeways, 26 bridges, 91 weirs and 1035 floodgates within the tidal reaches of coastal catchments in New South Wales.

Fragmentation by anthropogenic barriers has had negative impacts on endemic freshwater communities. In addition to changing important freshwater habitats (Harris, 1984), fragmentation has resulted in species homogenisation (Rolls, 2011), decline of populations, reduced species richness, high levels of larval mortality (Baumgartner *et al.*, 2006) and localised or complete species extinctions (Gehrke *et al.*, 2002; Brainwood *et al.*, 2008), to mention but a few. The impact of anthropogenic fragmentation of stream networks is well documented in the literature (e.g. Zwick, 1992; Jansson *et al.*, 2000; Fagan, 2002). However, the number of studies examining the patterns, cumulative impacts of multiple barriers and extent of fragmentation within entire riverine landscapes are limited (Thorp *et al.*, 2008).

Restoration of degraded riverine landscapes and their associated ecosystems, is important for their conservation both within Australia and worldwide (Lake, 2001; Lake, 2005). The value of goods and services provided by freshwater ecosystems and the threats to their persistence posed by human-induced pressures and alterations is widely recognised (Costanza *et al.*, 1997; Thoms and Sheldon, 2000; Baron *et al.*, 2002). The rivers, floodplains and wetlands of the Murray-Darling Basin, Australia, provides between $187 and $302 million worth of ecosystem services per annum and yet they are under significant pressure from widespread water resource development (Thoms and Sheldon, 2000). Substantial monetary investments are made annually on river restoration projects, with spending exceeding a billion dollars in the USA alone (Bernhardt *et al.*, 2005). Types of restoration projects include riparian management, in-stream habitat improvement, invasive species management, flow modification and dam removal (Bernhardt *et al.*, 2005). In parts of Victoria, Australia, the largest monetary investments into riverine restoration was spent on in-stream habitat improvement and riparian management projects between 1999 and 2001 (Brooks and Lake, 2007).

In the last decade, there has been a change in the focus of restoring ecosystems. There is a greater emphasis on restoring processes rather than structural patterns (Sear, 1994; Clarke *et al.*, 2003; Wohl *et al.*, 2005; Kondolf *et al.*, 2006; Beechie *et al.*, 2010). Restoration of pattern through activities such as improved in-stream habitat structure (e.g. boulder placement) has been criticised for various reasons, including the substantial variability in ecological response (e.g. Bond and Lake, 2003; Roni *et al.*, 2008; Palmer *et al.*, 2010). For example, restoring 13 lowland rivers in the United Kingdom through the installation of flow deflectors or artificial riffles did not result in significant changes in the abundance, richness and diversity of fish species in comparison to unrestored stream reaches (Pretty *et al.*, 2003). On the other hand, restoring riverine processes such as natural sediment and flow regimes is viewed as more desirable, because such projects are more self-sustainable in nature and embracing of the hierarchical spatiotemporal variability of riverine landscapes (Sear, 1994; Clarke *et al.*, 2003; Wohl *et al.*, 2005). Restoring riverine processes instead of patterns increases the likelihood of long term project success. Restoring connectivity is a means by which riverine processes can be facilitated. Riverine landscape connectivity has been improved through activities such as barrier removal (Catalano *et al.*, 2007), numerous structural alterations such as installation of fishways (Kowarsky and Ross, 1981), baffles (MacDonald and Davies, 2007) or low flow channels to existing barriers, and the improved management of flows through barriers (Boys *et al.*, 2012).

Stream networks are complex, hierarchically nested systems that operate at a variety of spatial and temporal scales (Frissell *et al.*, 1986; Amoros *et al.*, 1996). The drainage basin is the highest level of organisation in the fluvial hierarchy, extending over the largest spatial scale; while mesohabitats occupy the lowest level in the hierarchy and are exemplified by features such as homogeneous substrate patches, macrophytes or tree roots, occurring over small spatial scales (Petts and Amoros, 1996). Each level of organisation operates at a unique temporal scale being influenced by processes operating at different durations, magnitudes and frequencies. Long-term processes such as plate tectonics shape the drainage basin, while levels lower in the river hierarchy are governed by smaller-scale and higher-rate processes, with mesohabitats influenced by variation in flows (Petts and Amoros, 1996). The pattern of organisation at one level of organisation is constrained and influenced by its upper level, hence larger scale patterns and processes exert top-down constraints on the patterns and processes observed at lower levels. Because of these overarching catchment scale influences and the complexity of interactions between riverine processes, Bohn and Kershner (2002); Lake (2005); Wohl *et al.* (2005) and Roni *et al.* (2008) stress the importance of carrying out restorations within a 'basin' context.

Riverine restorations commonly occur, at-a-site, i.e., at smaller spatial scales (Bernhardt *et al.*, 2005; Alexander and Allan, 2006; Brooks and Lake, 2007). In the USA the average length of restored stream, per project, is 579 m (Sudduth *et al.*, 2007). Long-term ecosystem restoration initiatives targeting entire regions are limited and some of the more notable are the Florida Everglades (Koebel, 1995; CERP, 2013) and Chesapeake Bay (Hassett *et al.*, 2005) programs. Socioeconomic restrictions frequently limit the size of viable restoration projects (Wohl *et al.*, 2005). Because of these limitations, the importance of coordinating numerous smaller scale projects as part of larger catchment initiatives and quantifying their cumulative impacts has been identified (Lake, 2005). In addition, the location of smaller scale restorations is often opportunistically selected, irrespective of the need for restorations to be carried out within a catchment context. Alexander and Allan (2007) found that half of restoration projects that were examined in the states of Michigan, Ohio and Wisconsin, USA, reported ‘available land opportunities’ as the reason for site selection.

Methods have been developed to restore habitat connectivity within the context of the drainage basin. The importance of barriers for restoring habitat connectivity were prioritised with the use of score-and-rank techniques (e.g. Pethebridge *et al.*, 1998; Poplar-Jeffers *et al.*, 2009; Nunn and Cowx, 2012). Using these techniques, the importance of all barriers is judged and barriers are ranked in relation to structural and ecological criteria. However, these methods have been criticised for evaluating the importance of barriers independently and not incorporating the spatial configuration of these systems and the barriers within them when selecting restoration sites (O’Hanley and Tomberlin, 2005). Using optimisation approaches to prioritise barriers overcomes this problem and can result in significantly greater connectivity improvements compared to more traditional techniques (Kemp and O'Hanley, 2010). Furthermore, multiple barrier prioritisation methods developed from these approaches generally target habitat connectivity improvement for economically valuable fish species, such as salmon, which carry out anadromous migrations from oceans upstream into rivers to spawn. Thus these methods might not necessarily be the most applicable to Australian freshwater ecosystems, which are composed of numerous species with differing species behaviours (McDowall, 1996).

Optimisation is a branch of mathematics concerned with locating optimum solutions (either maxima or minima) to mathematical problems. Optimisation problems are generally composed of three key components, namely the objective function, decision variables and constraints (Pierre, 1969). Optimisation algorithms are used to solve and identify the optimum viable outcome to the respective problem within the specified constraints of the problem (Pierre, 1969). The development of the computer has had a significant influence on advancing the field of optimisation, enabling for larger and more complex problems to be modelled and investigated (Snyman, 2005).

Mathematical programming is highly applicable and desirable to assist with various world problems. It has found widespread application, especially in the field of engineering design, operations research and economics (Pardalos and Resende, 2002), where it is either applied to understand the system or aid managers and designers in optimal decision making. The transportation industry commonly uses models to optimise numerous operations related problems, such as scheduling of staff, scheduling of transport timetables and design of routes (Newman *et al.*, 2002; Yu and Thengvall, 2002). Furthermore, optimisation models also show potential for the efficient management of natural systems. Mathematical programming has been used for the protection of endangered species, for example, Larson *et al.* (2003) examined the likelihood of protecting piping plovers from predation in relation to the cost of the implemented protection strategy. Other applications of optimisation have also included waste management (ReVelle, 2000) and selection of reserve sites for biodiversity conservation in both terrestrial (Nicholson *et al.*, 2006; Westphal *et al.*, 2007) and marine (Sala *et al.*, 2002; Leslie *et al.*, 2003; Klein *et al.*, 2008) ecosystems.

More recently optimisation has been applied to the restoration of riverine landscapes. For example, Stralberg *et al.* (2009) applied an optimisation approach to the restoration of wetland habitat structure. In this study, the optimum distribution of habitat patches was determined so as to ensure that maximum ecological response was achieved for a number of differing bird species with differing habitat preferences (Stralberg *et al.*, 2009). Similarly, mathematical programming approaches have also been applied to identify the barriers to be removed to reconnect in-stream habitats for northern hemisphere anadromous species. Kuby *et al.* (2005) selected the optimum combination of dams for removal in relation to the dual objectives of maximising respective habitat gains while minimising losses in socioeconomic services provided by the selected dams. Similarly, O'Hanley and Tomberlin (2005) developed a model which optimally selected smaller barriers for removal within a budgetary limit to maximise restoration outcomes for salmonids.

Connectivity within Australian rivers has improved through the restoration of in-stream barriers (NSW Industry and Investment, 2009). Despite the importance of carrying out restorations within a catchment context being identified by Lake (2005), restoring connectivity of entire stream networks has not occurred in Australia. Restorations in the coastal catchments of eastern Australia have been attempted on a barrier-by-barrier basis only. Barriers were ranked and selected across multiple coastal catchments according to their potential individual ecological improvements. Applying an optimisation approach to guide barrier selections has proven highly valuable in reconnecting habitats within northern hemisphere catchments (Kuby *et al.*, 2005; O'Hanley and Tomberlin, 2005; Zheng *et al.*, 2009) and, could prove fruitful within an Australian context; although the use of such an approach to guide restorations is yet to be attempted in Australia.

This thesis adopts an optimisation-based approach to investigate the restoration of connectivity of an entire stream network. The Dendritic Connectivity Index (Cote *et al.*, 2009) is utilised to quantify the degree to which an entire stream network is connected and how restoration decisions can impact the level of connectivity. This thesis contributes to our understanding of river restoration at the scale of a drainage basin. In addition, it elucidates the influence of riverine landscape character on optimally restoring stream network connectivity. The thesis develops a series of methods to guide the optimal restoration of connectivity within an entire stream network. The approach taken is applied to the Hunter River catchment, New South Wales; a drainage basin where human activities have degraded the connectivity of the stream network (Harris, 1984; NSW DPI, 2006a; NSW DPI, 2006b).

1.2 RESEARCH AIMS AND QUESTIONS

The aim of this study was to determine an optimal-based solution for the restoration of longitudinal connectivity of a stream network heavily fragmented by anthropogenic barriers. This is achieved by addressing the following objectives:

Objective 1: What changes to the natural longitudinal connectivity of the Hunter River network have occurred because of human activities (Chapter 4).

Two research questions are posed:

- What are the abundance, nature and distribution of anthropogenic barriers in the Hunter River?
- Has the connectivity of the Hunter River stream network changed as a result of anthropogenic barriers?

Objective 2: To develop a predominantly desktop based approach, using Geographic Information Systems (GIS), for restoring longitudinal connectivity of entire stream networks (Chapters 5 and 6).

This objective develops an optimisation model to guide restoration decisions on which barriers are selected to be removed or modified, so as to achieve optimal improvements in stream network connectivity of the Hunter River. The model constructed, utilises remotely collected spatial data of the stream network and barriers. Research questions posed to address this objective are:

- How sensitive is the restoration of stream network connectivity to the character of the stream network and barrier characteristics?
- What impact does species behaviour have on the restoration of stream network connectivity?
- What influence does sequential decision making have on the overall optimal state of stream network connectivity?
- What are the critical barriers for restoring stream network connectivity of the Hunter River?
- How effective and representative is the use of the Dendritic Connectivity Index for optimisation of stream network connectivity restoration?

1.3 THESIS OUTLINE

The aim and objectives of this thesis are addressed in seven chapters. Following this introduction, Chapter two provides a literature review, which explores the characteristics of stream networks, and the role of connectivity in facilitating movement within them. A synthesis is also provided on the anthropogenic fragmentation of riverine connectivity; the extent of the problem and the consequent degradation of riverine environments. In addition, restoration of riverine landscapes is investigated, with particular emphasis on spatial scale and improvement of connectivity as a valuable restoration alternative. Finally, a summary of the use of optimisation as a tool to aid decision-making is provided, with particular attention to its current application in natural environments. Information on the study area is given in Chapter three. Detail on the geology, geomorphology, climate, hydrology, flow regulation, land use, vegetation and fish species of the region is provided. Chapter four examines the change in longitudinal connectivity of the Hunter River as a result of anthropogenic fragmentation. Emphasis on the impacts of different barrier types and the potential susceptibility of certain regions within the stream network to anthropogenic change is quantified. In Chapter five, an optimisation model is developed to assist in the optimal restoration of stream network connectivity, and its sensitivity to a number of stream, barrier, species and procedural characteristics is explored. This model is then applied to the longitudinal connectivity of the Hunter River in Chapter six through optimally targeting barriers which are most critical to the connectivity of the entire stream network. The final chapter is a synopsis of the study, reiterating the importance of scale in restoration and use of optimisation as a tool to restore connectivity and how the main findings of this study have addressed this. In addition, the implications of the findings to management are highlighted and suggestions for further research are made.

Chapter 2

2 Chapter 2 – A Literature Review

2.1 INTRODUCTION

This thesis investigates the potential restoration of longitudinal connectivity of an artificially fragmented stream network, the Hunter River, Australia. The approach is based on an interdisciplinary understanding of stream network connectivity; combining concepts of longitudinal hydrological connectivity in riverine landscapes and the spatial structure of stream networks. Particular attention is given to including all anthropogenic barriers affecting the stream network. Anthropogenic barriers are defined as any human constructed barrier in a river channel that influences process connectivity in the longitudinal dimension (Petts, 1984). The restoration of connectivity was undertaken at the scale of the entire stream network; thereby addressing an important knowledge gap identified at which restoration is undertaken. Optimisation methods are utilised to address this problem.

This chapter is made up of five main sections. The first reviews literature on conceptual models of riverine landscapes. The second reviews literature on stream networks and how they have been described, with particular emphasis on connectivity in river systems. The third reviews literature on the fragmentation of connectivity in riverine landscapes and its subsequent ecosystem impacts. The fourth section focuses on the field of river system restoration, highlighting the importance of scale at which restoration occurs and restoration in relation to longitudinal connectivity. The fifth section describes the components of optimisation modelling, its application to real world systems, and in particular the role of optimisation as a decision support tool for problems in natural environments. A synthesis of the literature leading to the overall approach undertaken in this thesis is provided in section six.

2.2 CONCEPTUAL MODELS OF RIVERINE LANDSCAPES

The importance of longitudinal connectivity for the structure and function of riverine landscapes has been acknowledged by many. The River Continuum Concept (RCC) of Vannote *et al.* (1980) was among the first of a multitude of conceptual models developed in the last three decades to deconstruct the complexities of river systems. The RCC portrays river system structure as a continuous gradient of changes in physical attributes (such as hydrology, channel proportions, sediment size distribution) and energy inputs from headwaters to river mouth. Vannote *et al.* (1980) advocate that this downstream gradient of changes in river morphology elicits a response in biotic community structure, which consequently adjusts in a predictable downstream fashion. This simplified model has been extensively criticised for a number of reasons, including its failure to recognise impacts of artificial impoundments on the river continua. Effects of river regulation were incorporated into the RCC by Ward and Stanford (1983) in their Serial Discontinuity Concept (SDC). The SDC hypothesises the downstream impact of dams on the river continua is governed, in part, upon their longitudinal position within the river network.

The importance of hydrologic connectivity in the lateral dimension is emphasised in the Flood Pulse Concept (FPC) of Junk *et al.* (1989). The FP emphasises the importance of overbank flows and the periodic exchanges of sediment and nutrients between the channel and floodplain to the functioning of aquatic ecosystems. While this model was initially restricted to flood pulsing in large tropical floodplain rivers it was extended by Tockner *et al.* (2000) to account for pulsing of flows in temperate regions. Ward and Stanford (1995) extended the SDC to include the impacts of flow regulation on rivers with lateral connections, arguing that impoundment of channels reduces the dynamics of flood pulses and decreases both the frequency and magnitude of overbank connectivity with floodplains, impacting the structure of both riparian and channel biota. In addition, the Riverine Productivity Model (RPM) was developed to account for the importance of in-stream primary production and riparian litterfall to food webs of large constrained rivers with negligible floodplains (Thorp and Delong, 1994).

All of these models (RCC, SDC, FPC and RPM) have been developed on the conceptual basis that river systems are a gradual continua, which is a paradigm that has recently received substantial criticism. Discontinuities are common in rivers and are exemplified through various features, including tributary junctions (Benda *et al.*, 2004), waterfalls and artificial impoundments (Ward and Stanford, 1983). A contrasting school of thought depicting river systems as hierarchically nested array of patches is gaining substantial support among river scientists (Frissell *et al.*, 1986; Poole, 2002; Wiens, 2002; Thorp *et al.*, 2008).

The Riverine Ecosystem Synthesis (RES) of Thorp *et al.* (2008) is the latest attempt to deconstruct the complexities of river systems. It is formulated on the concepts of eco-geomorphology (Thoms and Parsons, 2002) and landscape ecology, principally the theory of hierarchical patch dynamics (Wu and Loucks, 1995), and provides a framework for understanding river systems in terms of spatiotemporal variability. The RES advocates that the physical structure of rivers is not actually a continuous downstream gradient, but that river systems are rather made up of a collection of large discontinuous hydrogeomorphic patches, which are defined by a combination of the climate and catchment geomorphology, and may reoccur in a downstream fashion (Thorp *et al.*, 2008). The physical distinctness of the various hydrogeomorphic patches provides unique ecosystem habitats, which result in formation of inimitable ecological ‘functional process zones’ (FPZ).

2.3 STREAM NETWORKS

2.3.1 Networks

Networks are omnipresent features of the virtual and physical world. A common perspective of networks is they are composed of a set of nodes (or vertices) interconnected by edges (Newman, 2003). An example is a group of interconnected computers forming a local area network (LAN) in an office, school or university. Another view of networks is they are an interrelated set of corridors or pathways of the same type, where nodes occur at overlaps of two or more corridors (Forman, 1995). Examples of this include roads, hedgerows and rivers.

Flowing water erodes the surface of drainage basins and in doing so, forms an array of interconnected branching channels referred to as a stream network. Stream networks exist in a variety of spatial arrangements, including dendritic, trellis and parallel drainage patterns, where the spatial organisation is fundamentally controlled by the underlying geology and climate (Schumm and Lichty, 1965). Stream ordering, developed by Horton (1945) and later adjusted by Strahler (1957), is a common and widely applied method for describing and classifying stream networks. Headwater streams with no tributaries are classified as first order streams, with streams of second order originating at the confluence of two first order streams (Strahler, 1957). The meeting of two second order streams forms a third order stream and so on, with the highest stream order in the network being representative of the order of the drainage basin. In addition to stream ordering, numerous other statistical quantitative descriptors have been utilised to analyse stream network morphometry (Horton, 1945; Shreve, 1966), including drainage density, bifurcation ratios and stream-length ratios. Horton's (1945) two fundamental laws, the law of stream numbers and law of stream lengths, relate streams of differing orders within a drainage basin in terms of their abundance and lengths.

Stream networks provide the physical template upon which hydrological, geomorphological, and ecological processes occur (Southwood, 1977). Only 0.8 percent of the surface of the earth is covered with freshwater bodies (Gleick, 1996); yet freshwater is a fundamental resource for human societies, providing water for agriculture, power, navigation and a food resource. It is estimated that freshwaters provide habitat for approximately 100 000 surface water species (Hawksworth and Kalin-Arroyo, 1995) and between 50 000 and 100 000 groundwater species (Gibert and Deharveng, 2002). Thus understanding and quantifying the physical structure and character of stream networks is important for proper management of this landscape feature and critical resource.

2.3.2 Spatial Arrangement of Stream Networks

Riverine landscapes are complex interconnected systems but are associated with high levels of spatial and temporal variability in terms of their physical, chemical and ecological character. Riverine landscapes are viewed by many as hierarchically nested systems (Frissell *et al.*, 1986; Maddock, 1999; Thorp *et al.*, 2008) (Figure 2.1) where each level in the hierarchy operates at distinct spatial and temporal scales (O'Neill *et al.*, 1986; Wu and Loucks, 1995) and controlled by processes operating at distinct durations, magnitudes and frequencies (Petts and Amoros, 1996). The drainage basin is the highest level of organisation for investigating river systems (Chorley, 1967; Petts and Amoros, 1996; Thorp *et al.*, 2008). Drainage basins cover areas in the range of a few square kilometres to thousands of square kilometres and are delineated by the outlining watershed. Changes to the structure of entire drainage basins are brought about by large-scale and long-term processes, such as tectonic uplift, subsidence and glaciations (Frissell *et al.*, 1986), which can operate over many millennia. The network of channels within a drainage basin is composed of functional process zones, which represent a lower level in the hierarchical organisation of riverine landscapes. Functional process zones are characterised by a variety of channel types, such as anabranching, meandering or gorge sectors (Thorp *et al.*, 2008). At decreasing spatial and temporal levels or scales (Figure 2.1), the riverine landscape hierarchy is characterised by river reaches, functional sets, functional units and mesohabitats, which exist over smaller areas and are governed by processes acting over shorter time scales.

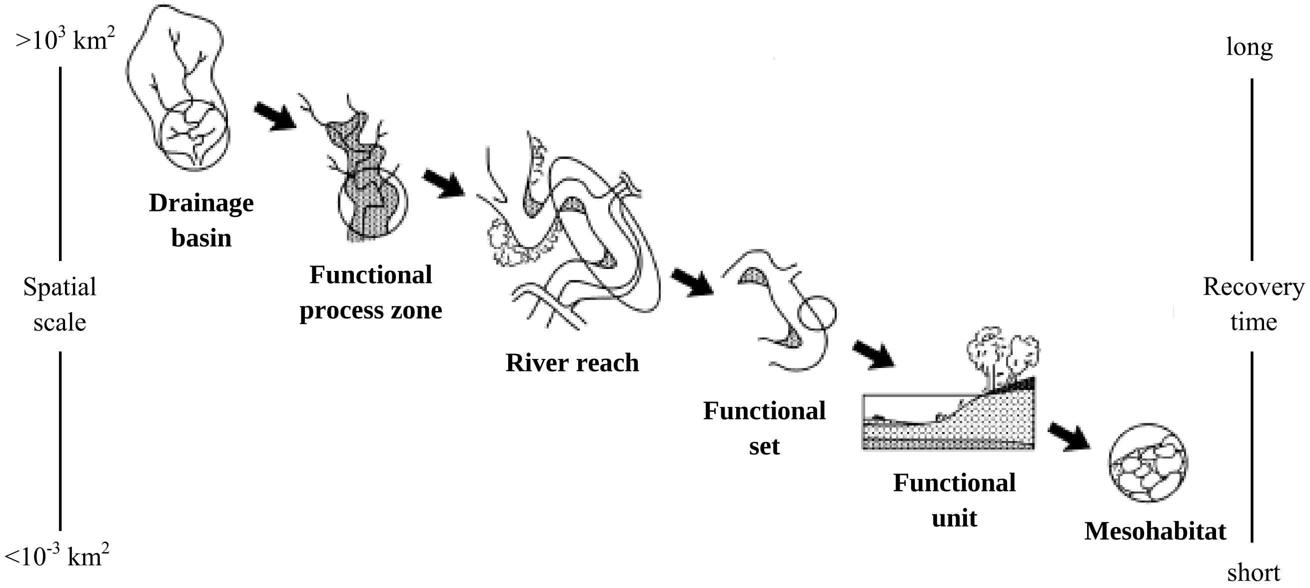


Figure 2.1 Hierarchical organisation of a riverine landscape (after Maddock, 1999; Thorp *et al.*, 2008).

Perceptions of the structure of riverine landscapes at the level of the drainage basin have been dominated by two contrary views. River systems have classically been depicted as clinal entities, where changes in sediment regime, hydrology, stream morphology and biotic composition occur predictably along a continuum from headwaters to mouth (Schumm, 1977; Vannote *et al.*, 1980). However, landscape features, such as tributary confluences, floodplains and varying geologies, ‘break’ this continuum. The influence of these features on the structure and function of riverine landscapes has been the subject of much debate (Poole, 2002; Thorp *et al.*, 2008), fuelling a contrasting conceptual basis for viewing rivers as a mosaic of distinct patches (Townsend, 1989; Montgomery, 1999; Rice *et al.*, 2001; Benda *et al.*, 2004; Thorp *et al.*, 2008). Additionally, the importance of incorporating the branched spatial configuration of patches into investigations of river systems has been highlighted, with branched structure and connectivity between patches having significant influences on ecological, hydrological and geomorphologic dynamics (Power and Dietrich, 2002; Ganio *et al.*, 2005; Campbell Grant *et al.*, 2007; Rodriguez-Iturbe *et al.*, 2009; Neeson *et al.*, 2011). For example, the spatial structure of stream networks is an important influence on fish species dispersal. In the Mid-Atlantic Highlands region of eastern USA, fish species richness is higher in streams that are in closer proximity to mainstem confluences than in streams that are more distant, while fish assemblages in headwater streams are influenced by local factors and not associated to stream network structure (Hitt and Angermeier, 2008).

2.3.3 Stream Network Connectivity

2.3.3.1 Connectivity

Connectivity is a central feature of natural and artificial systems, and is an important concept in geography, transportation theory, landscape ecology and riverine landscape studies (Forman, 1995). In ecological systems, connectivity between elements influences a multitude of processes, such as animal and seed dispersal, gene flow, colonisation of habitats, persistence of metapopulations and also propagation of disturbances (Turner *et al.*, 1989). Management of connectivity has been a central aim globally, for the long term conservation of ecosystem integrity and biodiversity.

Connectivity is defined as either structural or functional in nature (Tischendorf and Fahrig, 2000). Structural connectivity is concerned with the spatial arrangement and interactions between landscape elements upon which ecological processes occur (Collinge and Forman, 1998). For example, structural connectivity can refer to the arrangement of habitat patches within a landscape, or the configuration of tributaries making up a stream network. Functional connectivity was defined by Taylor *et al.* (1993) as ‘the degree to which the landscape facilitates or impedes movement among resource patches’, and as such is the interaction between landscape structure and the response of the object in question.

Movement between landscape elements generally is in relation to organisms; however understanding connectivity of other processes (e.g. materials, nutrients or energy) is also important, but less commonly investigated (Turner *et al.*, 2001). Landscapes are perceived differently by each organism, making connectivity a spatio-temporally variable characteristic of landscapes that are organism and scale-dependent (Crooks and Sanjayan, 2006). They are influenced by spatial attributes including patch quality, patch boundary, patch context and distance between patches (Wiens, 2002). Different disciplines have different perspectives on connectivity. Metapopulation ecologists perceive it as a patchwise characteristic, whereas landscape ecologists commonly associate it as a landscape attribute (Moilanen and Hanski, 2001), while conservationists relate it to habitat corridors (Pringle, 2003). River scientists generally associate it with water-mediated connections. However the notion of connectivity is perceived, it is accepted as being of critical importance to ecosystems.

2.3.3.2 Connectivity in Riverine Landscapes

2.3.3.2.1 Hydrological Connectivity in the Four Dimensions

River systems are described as the ‘epitome’ of connectivity (Wiens, 2002), because of the dominant influence exerted by the unidirectional flow of water in shaping and mediating patterns and processes within riverine landscapes. Hydrological connectivity was defined by Pringle (2001) as the ‘water-mediated transfer of matter, energy and/or organisms within or between elements of the hydrologic cycle’. Surface and groundwater flows therefore provide hydrological connections and exchange pathways between various riverine landscape elements (Amoros *et al.*, 1996) in four dimensions (Ward, 1989); longitudinal, lateral, vertical and overtime. Hydrologic connectivity in the longitudinal dimension refers to upstream-downstream exchanges, and can involve linkages between geomorphic units such as pools and riffles, or entire functional process zones. Laterally, the main river channel is connected to the adjacent floodplain during overbank inundation while in the vertical dimension flows occur between epigean waters and the hyporheic and phreatic zones (Thorp *et al.*, 2008). The degree of water-mediated connectivity is variable overtime, and, as such, is a function of the fourth dimension, time (Ward, 1989). Hydrological connectivity within riverine landscapes is critical to the functioning of freshwater ecosystems (Pringle, 2003).

2.3.3.2.2 Ecological Significance of Hydrological and Habitat Connectivity

Hydrological connectivity is important in the formation of freshwater ecosystem habitats. Flowing water facilitates vital ecosystem processes such as the transfer of organic matter and nutrients from riparian zones (Thorp and Delong, 1994), floodplains (Tockner *et al.*, 1999) and in-stream geomorphic features (Sheldon and Thoms, 2006), such as in-channel islands, into the river corridor. Similarly, water movement through the riverine landscape is associated with sediment erosion, transport and deposition which results in the formation of a variety of in-stream habitat features such as bars (Hooke, 2003). Overtime, sediment processes contribute to the physical spatial heterogeneity of riverine landscapes. Complex interactions between features of the riverine landscape and the intensity and nature of hydrological connections generate a mosaic of aquatic habitat types (Amoros and Bornette, 2002). Each aquatic habitat type has varying physical character such as substrate type, water velocities, flow depths, nutrient availability and degree of hydrological connectivity (Ebersole *et al.*, 2006). The availability of habitats provides freshwater organisms with areas for feeding, breeding, resting and cover from predators.

Hydrological connectivity facilitates the movement of organisms within the riverine landscape. River corridors which are formed during hydrological connectivity events provide dispersal pathways between resource patches (Junk *et al.*, 1989) and populations. Aquatic species, especially fish which are highly mobile, utilise a variety of habitats during recruitment, with the type and extent of life-cycle migrations varying between species. For example, Australian bass (*Macquaria novemaculeata*) have catadromous life-cycles, where females migrate from freshwater habitats downstream into brackish zones for spawning (Harris, 1986). Other species, such as golden perch (*Macquaria ambigua*) are potamodromous and undergo extensive upstream spawning migrations during high flows, covering distances in excess of 1000 km (Reynolds, 1983). In addition to the importance of surface water connectivity to life-cycle migrations of individuals, connectivity influences metapopulation fitness (gene flow) and the ability of metapopulations to recover from disturbances. Howell (2006) found that local fish populations fully recovered after extirpations associated with flood and fire disturbances, because connectivity with neighbouring populations was maintained.

The spatial configuration of riverine habitats and the respective connectivity between these are important for ecological process and pattern at different scales. Movement of biota can be restricted by biogeographic barriers or 'bottlenecks' at various spatial scales. Physical features on large spatial scales, such as oceans and mountain ranges, have been imperative in determining variations in regional fauna by restricting dispersal (Cox and Moore, 2010). Similarly, at smaller spatial scales, waterfalls and cascades can prevent upstream colonisation thereby promoting the development of distinct biotic communities. For example, McPhail and Lindsey (1986) found that fish species assemblages differed upstream and downstream of the 65m high Shoshone Falls on Snake River, Idaho. Three species were found only upstream of the falls, 18 only downstream of the falls and 12 species occurred on either side. Furthermore, two of the three species only occurring upstream of Shoshone Falls were also found in the neighbouring Bonneville Basin. Temporal hydrological variations exemplified through extreme flow events such as floods and droughts, impact on habitat connectivity, and consequently on ecological processes. Bonada *et al.* (2006) found that taxa richness of macroinvertebrates was lower in pool-riffle sequences with reduced longitudinal connectivity, while Bunn *et al.* (2006) highlight the tradeoffs of increased hydrological connectivity to biota in Australian dryland rivers.

2.3.3.2.3 Modelling and Measuring Connectivity within Stream Networks

Modelling and quantification of real world systems provide a means to predict and manage the responses of these systems, and also to investigate and gain further understanding of their character. Modelling of ecological patterns and processes within stream networks provides a method for understanding how connectivity affects ecosystems, and also presents a tool for informed resource management and conservation planning (Calabrese and Fagan, 2004).

The following section has three subsections. The first describes a number of models which have been developed to investigate the influence of stream network connectivity and structure on species dispersal and population dynamics. The second is a synopsis of the connectivity metrics which have been developed to succinctly quantify connectivity as an attribute of a habitat network. The final subsection is a brief overview on the 'dendritic connectivity index' (Cote *et al.*, 2009), which is employed as a measure of stream network connectivity in this thesis.

Models have been used to investigate the influence of stream network connectivity on population dynamics in relation to species dispersal. Fagan (2002) developed a hypothetical mathematical model to investigate the ramifications of habitat configuration (and changes in habitat connectivity brought upon by fragmentation) on metapopulation persistence, by modelling species dispersal through two network scenarios; a linearly and dendritically connected habitat network. Similarly, using a modelling-based approach, Labonne *et al.* (2008) evaluated the influence of branched stream network connectivity on metapopulation demographics, while Ganio *et al.* (2005) modelled the spatial distribution of cutthroat trout (*Oncorhynchus clarki clarki*) with explicit incorporation of stream network structure. Padgham and Webb (2010) developed a model to investigate the influence modifications to stream networks, in terms of habitat quality and connectivity between stream segments, had on the ease of mobility for fish through the network. Additional models have been constructed to explore the extent of colonisation of a network of stream-connected lakes in Sweden by pike (*Esox lucius*), in relation to dispersal ability between lakes (Spens *et al.*, 2007) and climate change (Hein *et al.*, 2011), both of which have direct application to the management of these inter-connected freshwater ecosystems.

Indices are commonly used to quantify and characterise the degree of connectivity in ecological systems. In terrestrial ecosystems, a plethora of indices have been developed to quantify both structural and functional landscape connectivity, from simple patch-based metrics such as nearest neighbour distance (Moilanen and Nieminen, 2002), to more complex landscape-scale metrics that incorporate both landscape and species attributes (Urban and Keitt, 2001). Detailed reviews on landscape connectivity indices are provided by Calabrese and Fagan (2004) and Pascual-Hortal and Saura (2006). However, indices for quantifying the spatial connectivity of stream networks are only beginning to be developed (e.g. Cote *et al.* 2009), with commonly used approaches in landscape ecology and network connectivity analyses, such as graph theory, only recently finding application in the study of riverine landscapes (Schick and Lindley, 2007; Erős *et al.*, 2011; Erős *et al.*, 2012).

The 'dendritic connectivity index' (DCI) of Cote et al (2009) is a basin-scale metric which quantifies the longitudinal connectivity of stream networks. In principle, the index measures connectivity based on the probability that an organism will be able to move between two random points in the network (Cote *et al.*, 2009), where a value of 100 indicates complete connectivity and lower values indicate reduced connectivity down to a minimum value of zero, which indicates total disconnection of the network. The index comes in two forms, the first is in relation to diadromous fish which migrate between freshwater and marine environments, and the second applies to fish with potamodromous species behaviour, thus only migrating within freshwaters. Both natural and artificial in-stream structures can act as obstacles to longitudinal mobility, thus decreasing stream network connectivity. The DCI is applicable to all stream networks with a branching stream configuration, and can also incorporate unique biological characteristics and hydrological variability.

2.4 ANTHROPOGENIC FRAGMENTATION OF STREAM NETWORK CONNECTIVITY

2.4.1 Landscape Fragmentation

Humans have utilised and altered natural environments for centuries. This has been done in multiple ways, including deforestation for resource extraction and land availability, to the building of extensive road networks and the construction of dams for water supply, flood protection and power generation. However, these actions have often been to the detriment of ecosystems, where habitats have been lost or highly degraded. In landscape ecology the term ‘fragmentation’ is synonymous with a landscape scale process resulting in habitat change (Forman, 1995; McIntyre and Hobbs, 1999), and is defined as the act or process of breaking something into fragments. It is often associated with decreased habitat connectivity (Fahrig, 2003), which has been labelled as a key driver of species persistence and ecosystem diversity (Fahrig and Merriam, 1985; Moilanen and Nieminen, 2002). Habitat fragmentation and the loss of connectivity corridors have been shown to have damaging effects of varying degrees on numerous terrestrial species, including plants (Jennersten, 1988), birds (Herkert, 1994), mammals (Verboom and Vanapeldoorn, 1990) and reptiles (Díaz *et al.*, 2000) to name a few. For example, forest fragments of the central Amazon rainforest generally had fewer species and sparser populations of dung and carrion beetles than intact areas, with minimal movement of beetles occurring into forest fragments (Klein, 1989).

2.4.2 Anthropogenic Fragmentation of Riverine Landscapes

Fragmentation is the consequence of multiple anthropogenic disturbances that have changed riverine landscapes, and has been labelled as a key driver of ecological degradation (Wohl, 2004). Humans have been altering riverine landscapes for a variety of purposes, such as flood control and protection, water security, reclamation of wetlands for agriculture and safe, fast passage across rivers. This has been done through stream channelization, the construction of dams, levees and road crossings, such as bridges, to mention a few. These structures have altered hydrological connections, and the associated transfer of sediments, nutrients, organic matter and biota, between components of the four-dimensional riverine landscape (refer to Section 2.3.3.2.1). For example, laterally, structures such as levees reduce overbank flows (Steinfeld and Kingsford, 2013), disrupting channel-floodplain exchanges and disconnecting species refugia (Ward and Stanford, 1995; Hohausová *et al.*, 2010). Vertically, siltation (Hancock, 2002) and channelization of streambeds fragment hyporheic zone processes. Upstream-downstream process linkages and critical pathways between ecosystem resource patches are fragmented by barriers such as dams. Changes in the flow regime reduce the dynamic nature of connections within the riverine landscape (Poff *et al.*, 1997).

Fragmentation of freshwater habitats and the reduction in connectivity between habitat patches is detrimental to biota – for two principal reasons. First, the branching structure of networks substantially limits the number of connections or dispersal pathways between habitat patches to aquatic species with a fully water bound life cycle, such as fish (Fagan, 2002; Campbell Grant *et al.*, 2007). A barrier can impede the limited dispersal routes between resource patches in a network which are necessary for life-stage completion, while increasing susceptibility to isolation of aquatic populations and reduced genetic diversity, with heightened possibility of extinction (Zwick, 1992). Second, anthropogenic barriers affect ecosystems indirectly by altering natural habitat forming processes, such as sediment and water flow, which in turn deteriorate habitat structure and function and influence aquatic biodiversity (Bunn and Arthington, 2002).

Anthropogenic fragmentation can have positive impacts on sustainability of native fish populations in some circumstances. The spread of exotic species and homogenization of fauna is seen as a significant and real threat to native ecosystems (Rahel, 2007). Artificial barriers can restrict the spread of exotic species into viable habitats and reduce external threats to native biota, preserving ecosystems. For example, in Australia, the presence of an impassable weir was used as a successful barrier to the upstream dispersal of exotic rainbow trout (*Oncorhynchus mykiss*) in the lower Cotter River catchment (Lintermans, 2000). This allowed for the recolonization of the upper stream reaches by the native species of mountain galaxias (*Galaxias olidus*), which were under threat due to predation from rainbow trout. In Norwegian streams, Holthe *et al.* (2005) investigated the possibility of placing species-selective artificial in-stream barriers. Such barriers were thought to reduce the spread of the exotic European minnow (*Phoxinus phoxinus*) into upper stream reaches, because the minnow has weaker jumping abilities than the native threatened brown trout (*Salmo trutta*).

2.4.3 Longitudinal Fragmentation by Anthropogenic Barriers and Impacts

Longitudinal connectivity within river networks has been fragmented by anthropogenic barriers. Common barrier types include water retaining structures, such as large dams, low-head dams and weirs, road crossings and floodgates. The impacts of individual structures on riverine landscapes, both worldwide and within Australia, has been reported in a vast array of literature. A descriptive review of common barrier types and their impact on river systems is provided below.

2.4.3.1 Water retaining structures

Water retaining structures such as dams have been long acknowledged to disrupt riverine processes longitudinally (Ward and Stanford, 1983). Dams act as a physical obstruction to water flow downstream, forming an upstream reservoir of highly lowered water velocities, changing stream reaches and associated freshwater habitats from lotic to lentic. Such structures can be small run-of-river farm impoundments which may have near insignificant effects on the flow regime, or mega-dams such as the Three Gorges Dam on the Yangtze River, China. Dams alter characteristics of in-stream habitat forming riverine processes and biotic exchanges, by reducing the downstream transfer of water, sediments (Walling, 2006) and nutrients (Caraco and Cole, 1999; Stanley and Doyle, 2002). Riparian plant community structure has also been found to have decreased continuity in impounded streams (Andersson *et al.*, 2000), with dams obstructing the natural water-mediated dispersal routes of seeds of certain plant species. For example, the distribution of riparian species with poor floating abilities was especially restricted (Jansson *et al.*, 2000), with in-water seed concentrations greatly reduced downstream of dams (Merritt and Wohl, 2006). Further, indirect impacts to riverine systems as a result of changes in the hydrological regime and resulting hydrological connections have been described by many, including Petts (1984), Poff *et al.* (1997) and Pringle *et al.* (2000a).

The direct impact of stream fragmentation by dams on aquatic species mobility and persistence is considerable. Dams prevent individuals from moving between critical habitats on either side of the barrier. The significance of reduced mobility to declining populations is well appreciated in the Americas (Kareiva *et al.*, 2000), Europe (Reyes-Gavilán *et al.*, 1996), Japan (Morita and Yamamoto, 2002) and Australia (Gehrke *et al.*, 2002), especially for highly migratory fish which need to move substantial distances to complete life cycles. Dam construction and reduced passage has been widely associated with various species disappearances, in addition to species homogenisation, declines in populations and reduced species richness (Joy and Death, 2001; Guenther and Spacie, 2006). For example, Tallowa Dam in NSW, Australia, drastically changed the once continuous lowland fish communities of the Shoalhaven River (Gehrke *et al.*, 2002). Differences between upstream and downstream communities include: reduced upstream species richness, localised extinctions of ten diadromous species, reduced abundance of four migratory species, downstream build up of juveniles and disappearance of Australian grayling (*Prototroctes maraena*) (Gehrke *et al.*, 2002). Effects of habitat fragmentation are not just restricted to highly mobile species. Impeded fish mobility has had consequences on freshwater mussels which are dependent on fish for dispersal. Brainwood *et al.* (2008) elucidated the possible functional extinction of mussel populations upstream of impoundments on the Hawkesbury-Nepean River, where, even though individuals are present, there is no evidence of recruitment.

The detrimental impact of barriers on aquatic organisms is not only restricted to large dams. In principle, low-head structures, such as weirs and milldams, can behave similarly to larger dams at low flows by reducing species access to critical resource patches and upstream spawning habitats (e.g. river lamprey *Lampetra fluviatilis*) (Lucas *et al.*, 2009). Aggregation of predatory species downstream of impassable weirs has resulted in alterations of trophic interactions due to the increase in competition (Baumgartner, 2007).

Low-head barriers such as weirs can also significantly compromise freshwater species recruitment and long-term sustainability. For example, larvae mortality of two iconic Australian freshwater fish, Murray cod (*Maccullochella peelii peelii*) and golden perch (*Macquaria ambigua*) was found to be particularly high when passing through a type of weir commonly found in Australian rivers (Baumgartner *et al.*, 2006). However, the degree to which low-head dams are impassable varies temporally, as an increase in stream flow can submerge the structure, making it passable to certain species, unlike with larger dams.

2.4.3.2 Floodgates

Connectivity in estuarine habitats and wetlands has been fragmented by flow restricting structures such as floodgates (Rogers *et al.*, 1994; Williams and Watford, 1997; Johnston *et al.*, 2003). A floodgate, also referred to as a flapgate or headwork, is defined as ‘a gate designed to exclude or reduce tidal flooding’ (Williams and Watford, 1997). Such barriers are commonly found in tidal reaches of Australian river channels (Johnston *et al.*, 2003). They span entire stream widths to prevent upstream tidal flows and flooding, but also simultaneously act as a physical barrier to passage of juvenile fish and invertebrates, restricting their access to critical life-stage feeding habitats and estuarine fish nurseries (Pollard and Hannan, 1994). In the Clarence River catchment, NSW, Australia, reaches upstream of floodgates had noticeably lower species biodiversity than connected reaches (Pollard and Hannan, 1994; Boys *et al.*, 2012). Species community structure was also altered, with significantly lower numbers of migratory biota (Pollard and Hannan, 1994; Kroon and Ansell, 2006). Concurrently, the reduced hydrologic dynamics within creeks due to floodgates was also detrimental to estuarine habitat and water quality (Kroon and Ansell, 2006).

2.4.3.3 Road crossings

Road crossings also act as longitudinal barriers to species mobility within river systems. Road crossings are structures which allow people to cross a river channel, and are present in a variety of forms, including bridges, culverts (water flows under the structure), causeways and fords which have water flowing over the structure. Such structures generally do not always act as physical barriers to stream flow like other barrier types. The potential for these barriers to fragment upstream species mobility is through changes in stream bed conditions, local hydraulics, such as increased flow velocities (Warren and Pardew, 1998), blockage and elevated barrier outfalls resulting from localised bed erosion (Park *et al.*, 2008). However, the degree to which the barrier hinders movement, known as barrier porosity, is variable and a function of the type of road crossing (Warren and Pardew, 1998), its physical condition, flow rate (MacDonald and Davies, 2007), the type of species (Nislow *et al.*, 2011) and their life stage (Burford *et al.*, 2009); as mobility characteristics are both species and life stage specific. Reaches upstream of impassable culverts have been found to have significantly reduced species abundance and richness of both fish (Nislow *et al.*, 2011) and crustaceans (Resh, 2005). Impassable road crossings can also facilitate the dispersal of exotic species by acting as a selective filter to native individuals with inferior mobility (Foster and Keller, 2011). Davis and Davis (2011) identified that increased flow velocities associated with culverts also impeded downstream movement of juvenile fish.

2.4.4 Fragmentation of Stream Networks

Stream networks have been extensively fragmented by anthropogenic barriers. Riverine fragmentation affects river networks worldwide, with approximately two thirds of the large river systems being impacted by large dam construction (Nilsson *et al.*, 2005). It is estimated that worldwide there are over 45 000 large dams which exceed 15m in height or with a reservoir capacity in excess of 3 000 ML (World Commission on Dams, 2000), while the number of smaller barriers is far greater (Poff and Hart, 2002). Walter and Merritts (2008) identified that in excess of 65 000 milldams were constructed just in the streams of the eastern United States. The Murray-Darling Basin, south-eastern Australia, is a large river system strongly affected by dams (Nilsson *et al.*, 2005), containing 3 600 weirs (Arthington and Pusey, 2003). Therefore, the need for the inclusion of small structures in stream network fragmentation analyses is growing (Garcia de Leaniz, 2008; Lucas *et al.*, 2009).

Concepts of habitat fragmentation from landscape ecology have been implemented in an attempt to understand changes in connectivity within stream networks (e.g. Fagan, 2002; Cote *et al.*, 2009; Erős *et al.*, 2012). Barriers ‘break’ stream networks into patches of habitat with reduced connectivity, impeding dispersal of aquatic species and potentially decreasing access to habitat. The degree of stream network fragmentation is partially determined by barrier location within the network (Rolls, 2011). For example, barriers located further downstream have more substantial effects on habitat availability to diadromous species than upstream barriers. Studies have investigated landscape scale anthropogenic fragmentation of stream networks and habitat loss to fish with anadromous migratory cycles (e.g. salmonoids) (Sheer and Steel, 2006; Poplar-Jeffers *et al.*, 2009). Park *et al.* (2008) found that up to 20 percent of stream habitat in Boreal forests was located upstream of impassable hanging culverts.

Fragmentation of stream networks and the high abundance of in-stream barriers have also been identified to influence stream geomorphology at a landscape scale. Walter and Merritts (2008) found that low order eastern USA streams were fragmented by more than 65 000 milldams by the mid-nineteenth century. This high abundance in barriers resulted in excessive sedimentation, which potentially transformed the riverine landscape from one characterised by anabranching channels to a region dominated by meandering streams.

The majority of studies on the fragmentation of stream networks have been carried out in the USA; mainly because of the economic and cultural significance of salmonoid species. However, Harris (1984) investigated stream network fragmentation in the coastal catchments of south-eastern Australia. New South Wales is the most heavily populated Australian state, with over 80 percent of its population located within 100km of the coast. As a result, the 22 coastal catchments in the region have been impacted by human disturbance and Harris (1984) estimates that over 11 000 km, or 32 percent, of stream habitat is impounded by 293 licensed artificial barriers. These estimates of stream impoundment and habitat fragmentation are conservative, as structures such as farm dams and road crossings, which are well known as potential barriers, were not included in this assessment.

2.5 STREAM NETWORK CONNECTIVITY RESTORATION

2.5.1 Introduction

Riverine landscapes and their associated ecosystems are some of the most human impacted and degraded natural environments in the world (Sala *et al.*, 2000; Millennium Ecosystem

Assessment, 2005). Extensive land use change, such as the conversion of forested landscapes or wetlands to agricultural land, has altered the structure and function, including increased sediment and nutrient loads, governing these systems. Riverine landscapes have also been further compromised through water resource development, river channelization, stream homogenisation and the construction of in-stream structures, all of which have reduced the dynamic nature and diversity of these systems. Restoration of highly degraded freshwater environments is a key concern for managers charged with conserving natural resources, not only because of growing environmental consciousness of vulnerable ecosystems, but also the threat to the multitude of critical ecosystem goods and services that riverine environments provide to society. Restoration of freshwater riverine environments encompass an assortment of strategies and projects, making up a multi-billion dollar industry across Europe, North America (Bernhardt *et al.*, 2005), Japan (Nakamura *et al.*, 2006) and Australia. In the Australian state of Victoria, it is estimated that approximately AU$130 million was spent on riverine restoration between 1999 and 2001 (Brooks and Lake, 2007).

To restore something is commonly defined as to 'return something to previous condition', such as a building or painting (Macquarie Dictionary, 2004). Bradshaw (1997) referred to environmental restoration as the process of returning the environment to its original state before artificial intervention occurred. However, in restoration ecology, the term 'restoration' is not clearly defined and is frequently interchangeably used with terms such as rehabilitation, remediation, reallocation and reconstruction (Hobbs and Norton, 1996). Henry and Amoros (1995) defined the restoration of natural environments, as 'returning an ecosystem to its conditions prior to disturbance (if known and possible), or, as in most cases, to a state as similar as possible to that which prevailed prior to disturbance'. While the goal of restoration has been described by Middleton (1999) as 'to establish a site that is self-regulating and integrated within its landscape, rather than to re-establish an aboriginal condition'. Hart *et al.* (2002) defined restoration as 'an effort to compensate for the negative effects of human activities on ecological systems by facilitating the establishment of natural components and regenerative processes'. Understanding of what degree and types of alterations to degraded environments constitutes as restoration is clearly ambiguous, especially because returning the environment to a natural state is an unlikely and unrealistic target. In this thesis, the term 'restoration' will be broadly used to refer to all human efforts and activities of improving the pattern or process (hydrological, geomorphological or ecological) of artificially degraded riverine landscapes, either wholly or partially. There is particular focus on any activities that constitute as ‘restoration’ of longitudinal connectivity in riverine landscapes.

2.5.2 Restoration of Riverine Landscapes

A plethora of human efforts have been undertaken in an attempt to restore pattern and process in degraded riverine landscapes (Bernhardt *et al.*, 2005; Brooks and Lake, 2007). Artificial disturbances both within the river channels and surrounding landscape have altered numerous components of riverine landscapes and ecosystems. The objectives and reasoning for restoration projects are highly variable; targets may be improvement of water quality or biodiversity conservation, while some are more socially motivated (aesthetic or recreational interests). Ecological restoration projects commonly target the improvement of aquatic habitat, through direct alterations to in-stream habitats (e.g. large wood placement, increases in habitat heterogeneity), channel reconfiguration, bank stabilisation (e.g. riprap installation) and riparian management (e.g. fencing, replanting and removal of exotic species). Many of these activities are based on the ‘field of dreams hypothesis’, which assumes that if habitat is restored so will ecosystem structure and function; however the success of such projects has been highly questioned (Bond and Lake, 2003; Palmer *et al.*, 2010). Less common approach to ecosystem restoration is via in-stream species management, which is carried out either through the introduction of threatened species (McNaught *et al.*, 1999) or extermination of exotic species which are detrimental to native ecosystems (Lintermans, 2000). Restoration of natural biotic and abiotic processes, through activities such as dam removal, floodplain reconnection, flow modification and fish passage improvement are also another means by which riverine environments have been restored. A synthesis on the type and number of riverine restoration projects undertaken in the USA and Victoria (Australia) is provided by Bernhardt *et al.* (2005) and Brooks and Lake (2007) respectively.

2.5.3 River Restoration and Scale

Traditionally riverine restoration efforts have been highly localised in terms of effort and commonly undertaken at small spatial scales (Clarke *et al.*, 2003; Thorp *et al.*, 2008). Both within Australia (Cottingham *et al.*, 2005; Brooks and Lake, 2007) and North America

(Bernhardt *et al.*, 2005; Alexander and Allan, 2006), river restoration has been dominated by reach or site-scale projects, with the median length of stream restored per project reported to be 579m in the USA (Sudduth *et al.*, 2007). Riparian management and in-stream habitat improvement have been the focus of implemented restoration activities (Bernhardt *et al.*, 2005; Alexander and Allan, 2006; Brooks and Lake, 2007; Sudduth *et al.*, 2007; Roni *et al.*, 2008). In addition to the limited understanding of riverine ecosystem functioning and restoration ecology, the relatively high costs of restoration and potential conflicts of interest surrounding the management of riverine resources restrict the physical scale of projects undertaken. However, in the last decade the number of larger scale ecosystem restoration efforts has increased, targeting regeneration of entire ecosystems like wetlands, estuaries and a focus on the reintroduction of natural flow regimes (Buijse *et al.*, 2002; Arthington and Pusey, 2003; Richardson *et al.*, 2005; Roni *et al.*, 2008).

Riverine landscapes are complex, hierarchically nested systems that function at a range of spatial and temporal scales (Frissell *et al.*, 1986). However, the importance of hierarchy and scale in the functioning of freshwater ecosystems is often limited in the implementation of river restoration projects. The success of projects has been variable (Roni *et al.*, 2008), and many restoration initiatives which specifically target improving ecological integrity and biodiversity have had limited biotic responses (Pretty *et al.*, 2003; Harrison *et al.*, 2004; Lepori *et al.*, 2005; Palmer *et al.*, 2010). This has been attributed to a mismatch in scale between the restoration activity and targeted outcome (Lake, 2001), where significant factors such as the grain and extent of processes and species are not incorporated into planning (Bond and Lake, 2003; Lake *et al.*, 2007). Furthermore, many have highlighted the importance of overarching catchment scale influences such as water quality, flow and sediment regimes on the failure of restoration activities (Sear, 1994; Thorp *et al.*, 2008). Yet, the location of restoration is generally chosen opportunistically (e.g. based on land availability) rather than specifically targeted within a larger context of catchment setting (Clarke *et al.*, 2003; Alexander and Allan, 2007), and subsequently coordination between multiple smaller scale projects is rare.

The importance of undertaking riverine restoration within a catchment scale context has been advocated by many (Bohn and Kershner, 2002; Lake, 2005; Lake *et al.*, 2007; Roni *et al.*, 2008). As stipulated by Wohl *et al.* (2005), it is more probable that restoration projects will succeed if carried out in the context of an entire catchment, because physical, biological and chemical processes have complex interactions across a variety of spatial and temporal scales. Sudduth *et al.* (2007) found that 51 percent of projects in southeastern USA were associated with a catchment scale assessment, but only 30 percent of restoration activities had been carried out as part of a larger scale plan. These projects were predominantly associated with water quality management, riparian management, channel reconfiguration and land acquisition. Consequently, scientists are highlighting the importance of coordinating multiple individual projects as part of a larger macro-project where cumulative catchment scale effects can be measured (Lake, 2005; Palmer and Bernhardt, 2006) because of the influence of financial restrictions on restoration size. For example, in addition to restoring environmental flows in the Snowy River, numerous other 'mini' projects are being carried out to improve the benefits of environmental flows, such as restoration of indigenous vegetation to floodplains (Lake, 2005). Systematic planning and prioritisation models represent tools which can be used in targeting location and types of projects for maximum restoration benefits (Thorp *et al.*, 2008; Hermoso *et al.*, 2012), especially as the extent of restoration required in a catchment can often substantially outweigh the available financial resources.

2.5.4 Connectivity Restoration

The restoration of connectivity within the riverine landscape is an important ecological restoration option. Riverine systems are also connected systems, with connectivity being a driving feature of the landscape, as highlighted in Section 2.3.3. Restoring connectivity facilitates riverine processes, which has been emphasised as a key aim and feature to successful river restoration (Wohl *et al.*, 2005; Kondolf *et al.*, 2006; Jansson *et al.*, 2007), while simultaneously embracing the self-sustaining nature of heterogeneous aquatic systems (Stanford *et al.*, 1996). Roni *et al.* (2002) ranked the reconnecting of aquatic habitats as the highest priority for restoration within degraded watersheds. Unlike other restoration foci, reconnecting critical components of riverine landscapes provides relatively fast response times, increased probability of project success, reduced variability in success between projects and significant project longevity (self-sustainability). Restoring connectivity also provides access to a diverse array of isolated habitats for entire ecosystems rather than focusing only on an individual attribute or specific species, often to the possible detriment of others (Roni *et al.*, 2002).

The literature on techniques for improving stream habitat connectivity, fragmented by anthropogenic barriers, is extensive. Connectivity is commonly improved through structural alterations, such as the installation of fishways (Kowarsky and Ross, 1981) or placement of baffles within culverts (MacDonald and Davies, 2007) to allow for fish passage. Changes in barrier operation is another procedure in which connectivity is improved, through the active management of tidal floodgates (Boys *et al.*, 2012) or with environmental flow releases for example. Barrier removal, is another, somewhat, more contentious connectivity restoration option which has been gaining increasing attention (Hart *et al.*, 2002; Poff and Hart, 2002; Stanley and Doyle, 2003). Barrier removal restores natural processes and also reconnects habitats which were previously isolated to aquatic fauna. The recovery of fish assemblages in upstream sites was found to be both extensive and rapid following low-head dam removal (Catalano *et al.*, 2007).

2.6 OPTIMISATION

2.6.1 Introduction

Optimisation relates to making something as perfect or effective as possible and involves locating an optimum. In mathematics, optimisation is concerned with choosing the best element from a set of elements (also referred to as decision variables), in relation to a certain quantifiable problem, and as such it has proven to be an indispensible tool in guiding decision making in real world scenarios (e.g. Kennedy, 1986; Lawrence Jr. and Pasternack, 1998). Generally, optimisation models are composed of three key components: the objective function, decision variables and the constraints. The objective function is a mathematical expression of whatever one is aiming to optimise, and it is a function of one’s decision variables. The decision variables represent what needs to be varied (and decided upon) to locate an optimal (maximum or minimum) outcome to an optimisation problem. Decision variables can be simultaneously restricted in value by a set of constraints, in order to make the solution feasible. The mathematical structure of both the objective function and constraints can vary in complexity from problem to problem, which is a fundamental factor in the classification of the type of optimisation problem. Different types of optimisation problems include linear, nonlinear, quadratic and integer programming. Various computational solution techniques (algorithms) have been specifically developed in order to solve the different optimisation problems of differing mathematical formulations. Dantzig's simplex method (Dantzig, 1963) is one of the earlier optimisation algorithms. Initially, developed during the Second World War as an aid to military operation, it is still one of the most commonly used algorithms for solving linear problems. A variety of heuristic techniques based on the natural world, such as that of genetic algorithms (Holland, 1975) and ant systems (Dorigo *et al.*, 1996) have proven to be highly useful and flexible in the solving of complex mathematical systems (Simpson *et al.*, 1994; Coley, 1999; Chen *et al.*, 2012). Genetic algorithms are based on Darwin's theories of evolution, with the concepts of natural selection, genetic crossover and mutation being the basis of the algorithm. Genetic algorithms have had broad applications, including in the fields of solid-state physics, facial recognition and water networks (Coley, 1999).

2.6.2 Applications of Mathematical Optimisation

The applicability of mathematical optimisation to real world systems is increasing, extending into areas of engineering design, operations research and economics. A boom in computing technology has facilitated widespread modelling of large and complex interconnected systems, allowing for optimal quantifiable decision making that would otherwise have been an extensive near-impossible task. Optimisation is therefore a highly desirable tool for managers. The literature on the application of optimisation and improvements in methodologies and algorithms is extensive, and a limited review will be provided as a general idea of its widespread use, especially as a tool for aiding decision making. Pardalos and Resende (2002) provide a wide synthesis on optimisation as an applied science.

Complex systems of significant social or economic importance are commonly formulated as optimisation models. As an example, the transportation industry (e.g. airlines (Yu and Thengvall, 2002), railways (Newman *et al.*, 2002)) commonly employ mathematical modelling for numerous tasks associated with their operations. Optimisation is implemented in the design of mechanical parts, it is also used in to schedule staff or formulate transportation routes and operation schedules. Optimisation also has extensive applications in the agricultural industry, where common problems include informed decision making on crop rotation for increased profit, or selection of the type and amount of food for cattle feed to ensure adequate nutritional intake at a minimum cost. Modelling is applicable in highly dynamic and uncertain systems, such as water resource management

(Loucks *et al.*, 2005), and also has potential in more static fields, including layout design of offshore gas pipeline systems or telecommunication networks. Other common areas in which optimisation models are implemented include computational molecular biology, production planning and the financial services industry.

2.6.3 Optimisation as a Tool for Conservation of the Natural Environment

Optimisation methods have been extensively applied to environmental systems. For example, mathematical optimisation as a tool for environmental quality management has been in use since the 1960s (Greenberg, 1995), where operations research methods have proven to be particularly valuable in management of waste, air pollution and water quality degradation (ReVelle, 2000). Similarly, optimisation models have also been used in the conservation of terrestrial (Nicholson *et al.*, 2006) and marine (Sala *et al.*, 2002) biodiversity. Computational methods for reserve site selection are extensively reviewed decision guiding tools (Pressey *et al.*, 1996; Arthur *et al.*, 1997; McDonnell *et al.*, 2002; Leslie *et al.*, 2003; Klein *et al.*, 2008), utilised in elucidating what reserve sites are most favourable for biodiversity conservation at a landscape scale. These reserve site selection methods have commonly been used in one of two ways (Cabeza and Moilanen, 2001); either to locate the smallest area of nature reserves required to sustain a certain predefined level of biodiversity (known as minimum area problem); or, to locate regions of highest biodiversity for a predefined size of nature reserve (maximal coverage problem). However, there are further variations in the model formulation which incorporate additional objectives, such as minimised fragmentation between selected reserves (Önal and Briers, 2005) or decreased boundary effect (Önal and Briers, 2003). Reserve network design has been applied in investigating conservation of plant (Virolainen *et al.*, 1999), bird (Westphal *et al.*, 2007) and invertebrate (Polasky *et al.*, 2001) biodiversity.

Mathematical programming has also been applied to guide decisions on conservation and recovery of endangered species populations. Larson *et al.* (2003) used a linear programming model to assess demographic and economic tradeoffs required to provide protection from predation for breeding pairs of piping plovers in the Great Plains region of the USA. This study optimised the likelihood of a successful response in population growth, in terms of the type of protection strategy that was implemented and the number of breeding pairs that would be protected by each strategy, where higher levels of protection were more costly but also had higher levels of success.

Optimisation models have also been used to guide restoration of terrestrial (Orsi *et al.*, 2011) and wetland (Newbold, 2005) environments. Crossman and Bryan (2006) developed a model to assist in prioritising the spatial location of habitat restoration decisions at a landscape scale. Their model chooses optimal regions for restoration in relation to maximising expected response of ecological benefits as a result of restoration, within the project budget. Stralberg *et al.* (2009) provide another example of mathematical modelling as a decision-aiding tool for wetland restoration. This model was used to decide whether, and which, habitat patches to restore from salt ponds to tidal marshes, in order to ensure maximum ecological sustenance of a number of bird species which have differing habitat preferences.

Optimisation models have been shown as a useful tool to guide the dam removal decision process. Kuby *et al.* (2005) developed a multi-objective optimisation model for prioritising dam removal, which incorporates both positive and negative tradeoffs of dam removal within a catchment. The approach taken attempts to maximise habitat availability for upstream migrating salmonoids by dam removal while minimising losses to socio-economic services (hydropower and water storage) provided by the set of dams. Even though the model elucidates the effectiveness of broad scale planning and systems analysis, it has been critiqued for firstly not incorporating monetary expenses of dam removal procedure, and secondly, for not allowing partial improvements to habitat availability without dam removal (Kemp and O'Hanley, 2010). More recently, Zheng *et al.* (2009) illustrated the use of multi-objective optimisation of dam removal prioritisation in a slightly different context. Optimal removal of dams was measured in relation to a number of consequent ecosystem response objectives, such as increases in fish abundance. Dam removal decisions were, however, constrained by the project budget, which was measured in relation to total cost of both barrier removal and exotic species control (as a result of increased habitat availability).

Similarly, optimisation models have proven to be successful and practical in steering management decisions on the removal of small in-stream structures (O’Hanley and Tomberlin, 2005; O’Hanley, 2011). An example of such a model is that of O’Hanley and

Tomberlin (2005). Their model selects optimal sets of small barriers (e.g. culverts) for removal within a project budget, so as to increase aquatic habitat availability in relation to fish with anadromous life cycles. Commonly, removal of such barriers has been prioritised through simple scoring-and-ranking systems. These systems generally judge the individual importance of each barrier in relation to certain criteria, such as structural (e.g. type of structure, ease of removal, barrier porosity, etc.) and ecological measures (e.g. amount of habitat upstream, location of habitat in the system, importance of habitat to endangered species) and then rank the structures in order of importance for restoration (cf. Pethebridge *et al.* (1998); Poplar-Jeffers *et al.* (2009); Nunn and Cowx (2012)). Even though scoring-and-ranking techniques are simple and straightforward to implement, significant limitations are that they do not account for barrier location within the wider catchment scale (Padgham and Webb, 2010) and interactions between multiple barriers, as they are assessed and ranked independently (Kemp and O'Hanley, 2010). Kemp and O'Hanley (2010) illustrate the superiority of optimisation methods as a decision aiding tool in such circumstances.

2.7 CHAPTER SUMMARY

Riverine landscapes are connected in four dimensions; longitudinal, lateral, vertical and overtime. At the scale of the drainage basin longitudinal connectivity is facilitated by a network of interconnected streams and river channels. Hydrological connectivity is an integral part of riverine landscapes, mediating extensive processes which are directly and indirectly critical to the functioning of ecological systems. Development and alterations have comprised the ecological integrity of riverine landscapes, where the loss of connectivity by anthropogenic fragmentation has disrupted the movement of sediment, water and nutrients, in addition to decreased habitat availability and restriction on dispersal of species. Anthropogenic fragmentation is a significant and widespread threat to riverine landscapes. Freshwater ecosystems are amongst the most degraded and threatened of global ecosystems. Restorations are being implemented in an attempt to remediate the ecological and functional condition of these resources. The need for restoration to be done within a drainage basin context has been reiterated (Bohn and Kershner, 2002; Wohl *et al.*, 2005; Roni *et al.*, 2008), where the use of optimisation as a tool in systematic planning and as a guide in decision making is showing promise in multiple environmental contexts. As such, the primary aim of this thesis is to optimise network connectivity restoration in an artificially fragmented river system, so as to maximise the ecological outcomes of monetary investments at a catchment scale. This thesis investigates the extent to which barriers have potentially fragmented the Hunter River catchment, so as to gain a further understanding of potential threats to stream network connectivity.

Chapter 3

3 Chapter 3 – Study Area

3.1 INTRODUCTION

The Hunter River Basin, New South Wales, S.E. Australia, is the study area for the research undertaken in this thesis (Figure 3.1). It is the furthest inland extending coastal catchment in New South Wales and has a catchment area of 22 000 km^2 (Chessman *et al.*, 1997). The Hunter River catchment is bounded by the Murray-Darling Basin to the west, the Manning and Karuah Basins to the north and the Hawkesbury and Macquarie-Tuggerah Lakes Basin to the south.

The headwaters of the Hunter River rise in the Mount Royal Range in the north-east of the catchment (Figure 3.2). The Hunter River flows in a broadly easterly direction to reach the Pacific Ocean approximately 120 km north of Sydney. The river initially flows in a south-westerly direction (Figure 3.1). The Goulburn River, its first major tributary, meets the Upper Hunter from the west, forming the easterly-flowing Hunter River. Further downstream Wollombi Brook joins the Hunter River from the south, after which the Paterson and Williams Rivers meet it from the north. After flowing for 452.5 km the Hunter River discharges at Newcastle into the Pacific Ocean. Geology, geomorphology, climate, hydrology and vegetation all factors that have shaped the catchment and associated river systems as well as land use are discussed in this chapter.

3.2 GEOLOGY AND GEOMORPHOLOGY

The Hunter catchment has four major geologic provinces: the New England Geosyncline, Sydney Basin, Great Artesian Basin and East Australian Tertiary Volcanic Province (Galloway, 1963b). The complex geology of the Hunter catchment has resulted in a highly varied landscape, composed of contrasting areas of mountainous terrain to flat lowland valleys. The Hunter catchment is divided by a large fault system, the Hunter-Mooki Thrust system, which separates the Carboniferous and Devonian sediments located in the north-

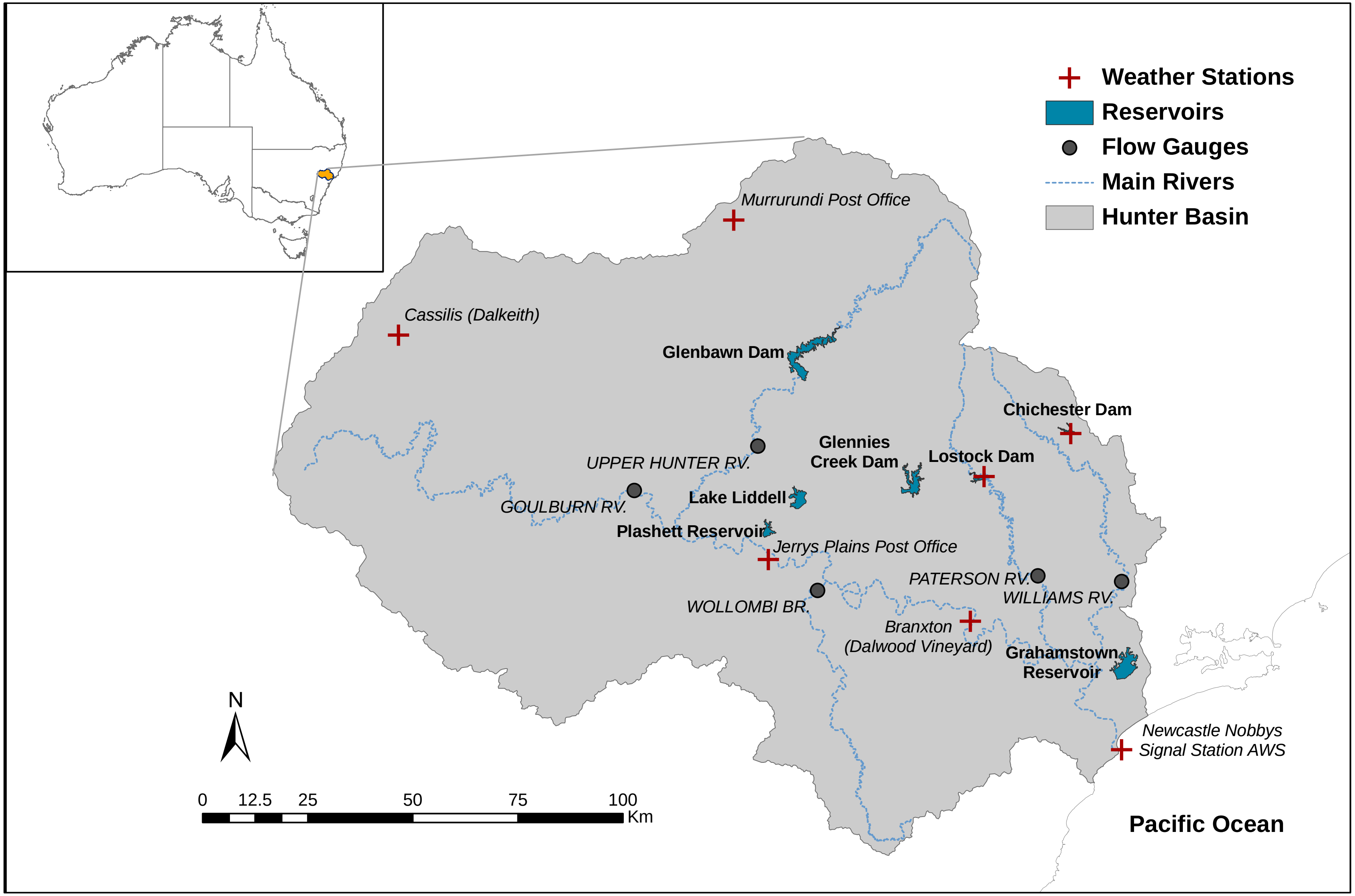


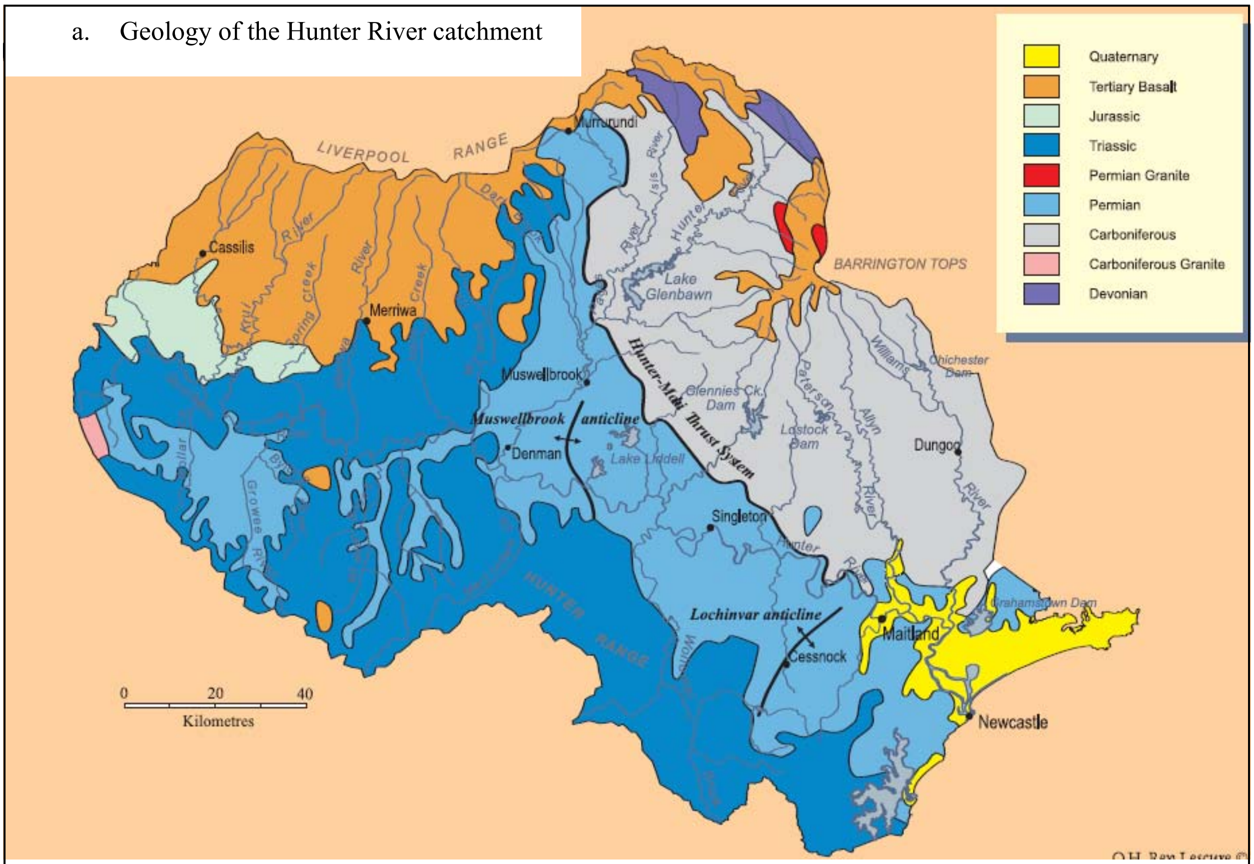
Figure 3.1 The major rivers and reservoirs of the Hunter River catchment, with the location of flow gauges and weather stations indicated.

eastern sections of the catchment from the large deposits of Permian sediments and Triassic sandstones and conglomerates found south and west of the fault line (Figure 3.2a). In addition, a further three major geologic structures occur in the catchment, namely the Tertiary basalts, Jurassic sandstones and conglomerates, and the Quaternary alluvium deposits and coastal mudflats (Peake, 2003; Figure 3.2a).

The mountainous north-eastern regions of the catchment extending to the Barrington Tops are dominated by Permian granites and highly-resistant and folded marine Devonian and Carboniferous sediments, composed of siltstone, sandstone and chert (Galloway, 1963b). Minor deposits of Jurassic sandstone and exposed igneous intrusions are also evident in the western region around Cassilis and north-east of the catchment respectively. Resistant Triassic sediments, primarily composed of sandstone, shale and conglomerates, cover the large rugged southern and western landscapes of the Hunter. These Triassic sediments form shallow soil of poor quality (Galloway, 1963b). The central valley floor of the Hunter River, extending between Murrurundi and Newcastle, is made up of less resistant Permian sediments which have been eroded to form undulating hills and lowlands. These sediments are composed of shale, tuffs, sandstone, conglomerates and coal deposits, and have originated from marine and estuarine sediments (DLWC, 2000). As a result of their marine origins, the sediments have a high salt content, resulting in naturally high salinity levels in the surrounding rivers and groundwater (DLWC, 2000). During the most recent Quaternary period, a complex sequence of erosion and deposition events have occurred in lower parts of the catchment, because of drastic climatic and sea level changes (Galloway, 1963b). This has resulted in the formation of unconsolidated deposits of alluvium along the lower reaches of the Hunter River and surrounding creeks, making up some of the highest quality agricultural land in the region (Peake, 2003).

The regional geology and geomorphology of the Hunter catchment are closely associated (Galloway, 1963a; Figure 3.2). The Liverpool, Mount Royal and the Barrington Tops mountain ranges form the northern divide of the catchment and they are characterised by a combination of deep valleys, ridge crests and high plateaux (Figure 3.2). Rugged zones of shallow soils, steep ridges and narrow gorges carved by headwater streams characterise the upper regions of the north-eastern mountains. Steep, large hills and plateaux occur on the southern and western extents of the north-eastern mountains. Southern-flowing streams dissect the Merriwa Plateau, forming deep valleys in parts of the generally hilly landscape.

a. Geology of the Hunter River catchment



b. Geomorphology of the Hunter River catchment

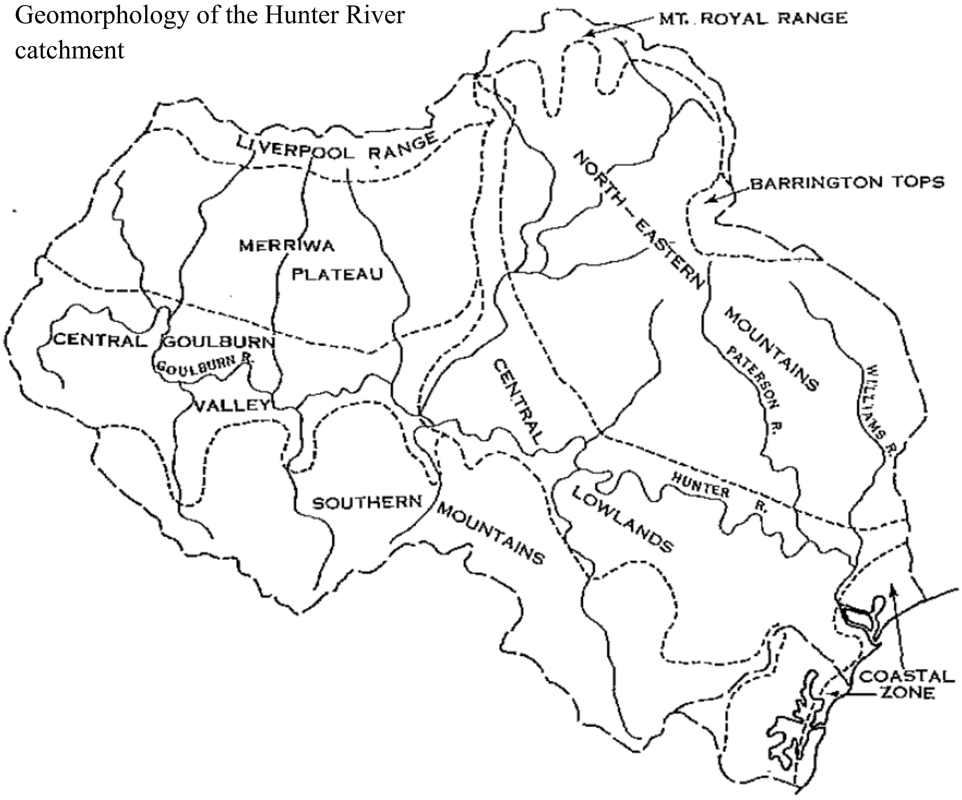


Figure 3.2 Major a). Geologic, and b). Geomorphologic regions of the Hunter River catchment, after Schnelder (2007) and Galloway (1963a).

Rugged mountains, sandstone plateaux and ridges broken up by steep valleys characterise the southern mountains and some parts of the Central Goulburn Valley, where the Goulburn River passes through steep gorges. The central lowlands are made up of undulating country, characterised by widely spaced shallow valleys and mild topographic relief.

The river bed sediments of the Hunter River are closely associated with the regional geology. Streams in the southern region, such as Wollombi Brook and Goulburn River, flow through Triassic sediments and their channel beds are made up of sand. Thus, the nature of sediments in the lower Hunter River is predominantly sandy. Streams in the north and north-east of the catchment have beds composed of gravel, cobbles and boulders, due to the highly resistant underlying Carboniferous rocks and Tertiary basalts. The Williams River, located on the north-eastern mountains, is a steep gravel bed stream, where inchannel benches and bars are common geomorphic forms (Erskine, 2001). The river alternates between reaches of close bedrock confinement and unconfined floodplains (Erskine, 2001), with upstream sections exhibiting a trellis drainage pattern. It is representative of rivers located in the north-eastern mountains. The sub-network of streams on the Merriwa plateau flow in a southerly direction and exhibit a parallel drainage pattern (Figure 3.2a), while the rest of the network portrays a dendritic drainage pattern.

The Hunter River is an order six stream according to the Strahler method applied to a 1:100 000 scale map, and has a variable geomorphology along its length. In the upper reaches, it is a steep low sinuosity single channel confined by the underlying bedrock (Fryirs *et al.*, 2009). This is followed downstream by a partly-confined section, after which the river moves from a hilly landscape to undulating plains, becoming a meandering laterally unconfined stream with low to moderate sinuosity (Fryirs *et al.*, 2009). The river bed is composed of gravel sediments and characterised by pool-riffle sequences. The width of the channel here varies between 10 m and 70 m (Erskine, 1992; Hoyle *et al.*, 2008), while the macrochannel is in the range of 75 m to 600 m (Hoyle *et al.*, 2008). Downstream of the confluence with the Goulburn River, the nature of Hunter River bed sediments changes to sand-dominated (Hoyle *et al.*, 2008). Alluvial flats extend around the lower Hunter River and its tributaries, forming wide floodplains through which the highly sinuous river channel meanders. However, some reaches of the lower Hunter River have undergone significant planform alterations that have reduced river sinuosity from 3.84 to

1.38 (Erskine and Warner, 1988). At Maitland, the river has an approximate bankfull width of 180 m (Erskine and Warner, 1988).

3.3 CLIMATE

Majority of the Hunter catchment has a warm temperate climate (Chessman *et al.*, 1997). There is a marked rainfall gradient from east to west influenced by the ocean (Bridgman, 1984) with long term mean annual rainfall (1862-2012) declining from 1134 mm at Newcastle to 624 mm at Cassilis (Table 3.1, Figure 3.1). Topography also has a dominant influence on regional rainfall, with highest mean annual rainfall of 1321 mm occurring in the mountainous regions of the Barrington Tops (Table 3.1, Chichester Dam) where rain-bearing south-easterly winds are intercepted (Bridgman, 1984). Rainfall is dominated by wetter and drier periods. The interior receives most of its rainfall during the summer months, between October and March (Table 3.1), while the coastal region is dominated by autumn and winter showers (February to June), due to the incoming high-in-moisture Tasman air masses (Tweedie, 1963). Over a longer time scale, climate in the catchment is characterised by periods of sustained higher or lower rainfall occurring over several decades, known as flood and drought-dominated regimes respectively (Erskine and Warner, 1988). Flood-dominated regimes have occurred from 1857 to 1900, 1949 to 1990, while drought-dominated regimes from 1821 to 1856, 1901 to 1948 and 1991 to 2008 (Erskine and Warner, 1988; Erskine and Townley-Jones, 2009).

Mean maximum monthly temperatures throughout the Hunter catchment are relatively uniform. Summer months are influenced by synoptic highs (Bridgman, 1984); January being the hottest month, with mean maximum temperatures reaching around 30ºC across the catchment (Table 3.1). During summer, commonly occurring low pressure troughs produce coastal sea breezes influencing coastal temperatures (Bridgman, 1984), with mean maximums in Newcastle (25.5ºC) being cooler than across the rest of the catchment. Winter months are dominated by southern maritime air masses (Tweedie, 1963), bringing high risks of frost in the central and western regions and the common occurrence of snow in the northern mountain ranges during this period (Bridgman, 1984; Chessman *et al.*, 1997). July is the coolest month in the Hunter region with maximums of about 16ºC across the catchment (Table 3.1). Local variations in climate are significantly influenced by

Table 3.1 Mean monthly rainfall (mm), maximum and minimum temperatures (ºC) at selected sites in the Hunter catchment (Bureau of Meteorology, 2012). Rainfall records are from 1874-2012 and temperature records are from 1907-1966 for Cassilis, 1884-2012 and 1907-2011 for Jerrys Plain Post Office, and 1870-2012 and 1907-2010 for Murrurundi Post Office. Rainfall and temperature records are from 1969-2012 for Lostock Dam and 1882-2012 for Newcastle Nobbys Signal Station. Rainfall records are from 1942-2012 for Chichester Dam and 1863 to 2013 for Branxton. Locations of all weather stations indicated in Figure 3.1.

	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Annual**
Cassilis (Dalkeith), Latitude: 32.00°S, Longitude: 149.99°E, 420m													
Rainfall (mm)	72.4	68.5	53.8	41	39.9	46.3	42.4	43.9	44.4	50.7	55.9	64.7	***624.2***
Max T (°C)	29.5	28.3	26.1	21.8	17.9	14.6	14.1	15.9	19.6	23.2	26.4	28.8	***22.2***
Min T (°C)	16	15.7	13.5	9.2	5.3	3.2	1.9	2.8	5.1	8.7	11.9	14.8	***9***
Jerrys Plains Post Office, Latitude: 32.50°S, Longitude: 150.91°E, 90m													
Rainfall (mm)	76.7	72.8	58.4	44.5	40.9	48.1	43.5	36.5	42	52.2	61.1	67.9	***645.2***
Max T (°C)	31.7	30.9	28.9	25.3	21.3	18	17.4	19.4	22.9	26.2	29.1	31.3	***25.2***
Min T (°C)	17.1	17.1	15	11	7.5	5.3	3.8	4.4	7	10.3	13.2	15.7	***10.6***
Lostock Dam, Latitude: 32.33°S, Longitude: 151.46°E, 200m													
Rainfall (mm)	125.7	128.2	120.8	71.9	75.7	67.7	39.8	36	50.8	68.2	90.7	93.8	***971.1***
Max T (°C)	29.3	28.2	26.5	23.5	19.9	16.9	16.4	18.4	21.5	24.5	26.3	28.8	***23.4***
Min T (°C)	17.3	17.2	15.4	12.6	10	7.7	6.5	6.9	9.3	11.9	13.9	16.1	***12.1***
Murrurundi Post Office, Latitude: 31.77°S, Longitude: 150.84°E, 466m													
Rainfall (mm)	90.8	78.8	61.2	52.9	54.8	69.8	63.6	61.8	58.3	73.2	75.7	89.8	***831.2***
Max T (°C)	30.8	29.8	27.8	23.7	19.2	15.8	15.1	16.9	20.6	24.1	27.3	29.7	***23.4***
Min T (°C)	15.2	15	12.6	8.5	5.1	3.2	2	2.5	5	8.2	11.1	13.6	***8.5***
Newcastle Nobbys Signal Station AWS, Latitude: 32.92°S, Longitude: 151.80°E, 33m													
Rainfall (mm)	88.1	107.7	119.7	116.1	118	117.1	94.7	73.7	72.9	73.3	70.7	81	***1134.3***
Max T (°C)	25.5	25.4	24.7	22.8	20	17.5	16.7	18	20.2	22.1	23.5	24.9	***21.8***
Min T (°C)	19.2	19.3	18.3	15.3	12	9.7	8.4	9.2	11.4	14	16.1	18	***14.2***
Chichester Dam, Latitude: 32.24°S, Longitude: 151.68°E, 194m													
Rainfall (mm)	164.1	183.1	170.2	100.1	96.9	104	54.1	60.3	63.1	93.3	105.3	125.2	***1320.8***
Branxton (Dalwood Vineyard), Latitude: 32.64ºS, Longitude: 151.42ºE, 40m													
Rainfall (mm)	87.0	95.3	90.3	68.4	61.0	68.4	51.8	44.9	50.2	58.9	63.4	78.2	***813.3***

oceanic effects. Mean monthly minimum temperatures are characterised by a decreasing gradient from the east to the west of the catchment where temperatures are consistently the highest at Newcastle irrespective of the time of year (Table 3.1). In the warm months, minima vary between 15ºC at Murrurundi and 20ºC at Newcastle; while during the cool months mean temperatures drop to 8.4ºC in Newcastle and to less than 2ºC in Cassilis (Table 3.1).

3.4 HYDROLOGY AND FLOW REGULATION

The hydrology of the Hunter catchment is both spatially and temporally variable. The mean annual discharge of the Hunter catchment is 1 680 000 ML (Chessman *et al.*, 1997). The distribution of flows is highly influenced by the seasonality and decreasing east-west rainfall gradient. Most of the water in the Hunter River is supplied from the north-easterly high rainfall region of the catchment by the Williams and Paterson Rivers, which have mean annual discharges of 335 949 ML and 239 986 ML respectively (Table 3.2). These are the few rivers in the catchment that have consistent, perennial flows (Chessman *et al.*, 1997) and are characterised by high peak annual discharges and low variability in flows (Table 3.2). On the other hand, the Goulburn River and Wollombi Brook in the west and south have substantially lower mean annual discharges of 176 328 ML and 140 221 ML, even though they drain significantly larger areas of the catchment (Table 3.2). The flow in these rivers is highly variable (Table 3.2) and marked with periods of low or no flow, low to moderate peak annual discharges and very high flood variability (Erskine and Livingstone, 1999). Seasonality in river flows throughout the catchment is reflected by the summer-dominated seasonality in rainfall. High flows generally occur between January and June, with local flow peaks occurring predominantly in February/March with a secondary peak in June (Figure 3.3). The low flow season extends from July until December, when flow ceases in numerous creeks. However, seasonality of flow in the Upper Hunter River is inverted with flow peaks occurring in July (Figure 3.3).

Long term variability in hydrology of the Hunter catchment is characterised by alternating multi-decadal periods of flood (FDR) and drought dominated regimes (DDR) (Erskine and Warner, 1988). Periods of increased flood frequency have occurred from 1857 to 1900 and since 1949, coinciding with periods of higher rainfall in the Hunter catchment (Section

Table 3.2 Flow statistics of major tributaries within the Hunter catchment (Source: NSW Department of Water (Online)). Location of all gauges indicated on Figure 3.1.

		River				
		Upper Hunter	***Goulburn***	***Wollombi***	***Paterson***	***Williams***
Gauging Details	***Gauge Number***	210002	210031	210004	210079	210010
	Catchment Area (km^2)	4220	6810	1848	956	997.5
	Period of Record	1907-2012	1954-2012	1908-2012	1928-2012	1927-2012
Annual Discharge Data (ML)	***Mean***	350 080	176 328	140 221	239 986	335 949
	Mean/km2	83.0	25.9	75.9	251.0	336.8
	Max	1 080 710	1 247 749	950 095	579 540	1 135 489
	Min	73 063	0	51	13 559	6 936
	CV	0.80	1.21	1.22	0.61	0.71
Mean Daily Discharge Exceedance Probability (ML/Day)	***20***	738.8	362.3	173.2	511.3	625
	50	358.4	102.8	37.9	134.6	126.1
	80	162.2	11.03	2.5	45.2	30.9
Peak Annual Discharge (ML/Day)	***Mean***	27 541	25 408	22 615	29 908	43 722
	CV	1.31	1.67	1.85	0.88	0.80
Max Daily Discharge on Record (ML)		117 240	282 500	262 820	73 208	119 750

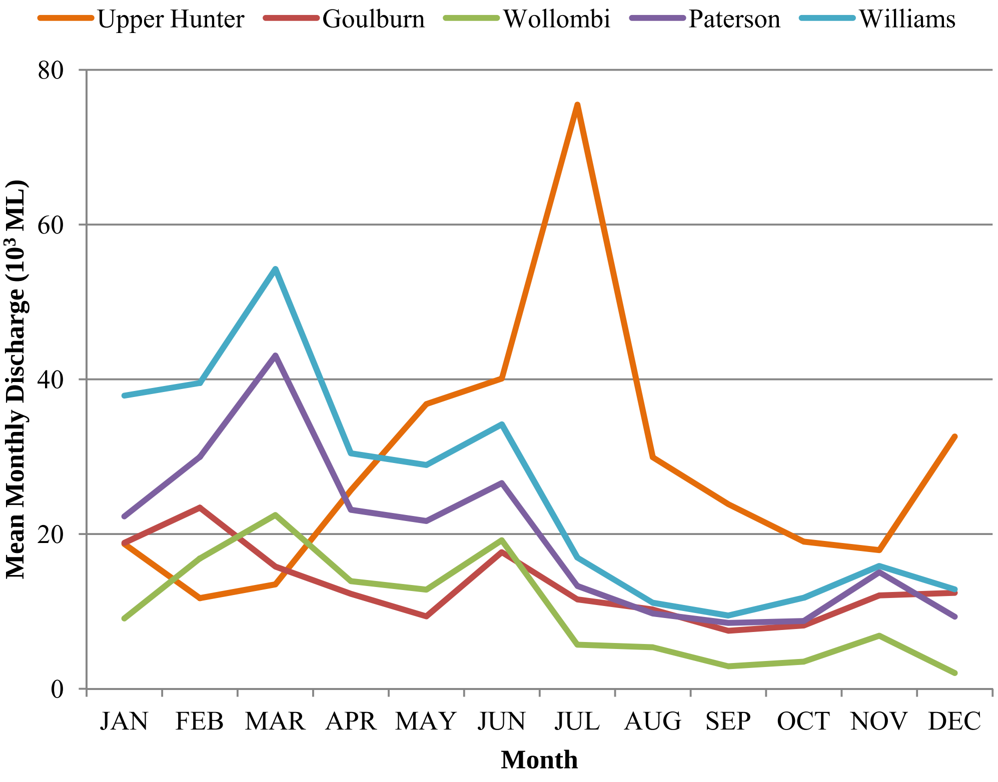


Figure 3.3 Mean monthly discharge at each river gauging station (Upper Hunter 1913 to 1927, with all other rivers 1969 to 2012) (Source: NSW Department of Water (Online)).

3.3). The flood of 1955 is the largest in the Hunter River since European settlement, with a discharge of 5685 m^3/s and a recurrence interval of one in 100 years in the Upper Hunter River (Hoyle *et al.*, 2008). Flood activity in the Hunter catchment is highly variable in relation to both Australian and international standards (McMahon *et al.*, 1992; Erskine and Saynor, 1996). Flood variability in Wollombi Brook and Goulburn River is the highest in the Hunter catchment and substantially larger than in the more hydrologically stable Williams River (Erskine and Saynor, 1996; Erskine, 2001).

Water resource developments in the Hunter catchment started in the early 20th century. Flows are intercepted by eight large water retaining structures, seven of which are dams (Table 3.3) and one a weir (Figure 3.1). Glenbawn Dam, located on the upper Hunter River, is the largest reservoir in the catchment with a storage capacity of 870 000 ML (100m high wall). The dam was constructed in 1958 (and trebled in volume in 1987) and acts as a dual purpose reservoir, providing flood protection while simultaneously supplying water to downstream areas (Erskine, 1985) predominantly for irrigation purposes. Flows in the Hunter River are further regulated by an additional two reservoirs, Lostock (20 000 ML) and Glennies Creek Dams (283 000 ML). Lake Liddell (148 000 ML) and Plashett Reservoir (65 000 ML) are utilised as cooling storages to the nearby Liddell and Bayswater power stations (NSW Department of Water, Online). Urban water to Newcastle and the surrounding regions is supplied by Chichester Dam (22 000 ML) and Grahamstown Reservoir (132 000 ML) (Erskine, 2001). Water from the Williams River is diverted to the latter by Seaham Weir, a 2.5m high, fixed crest weir located on the Williams River 14.9 km upstream from its intersection with the Hunter River (Rolls, 2011). Furthermore, the weir acts to limit the upstream movement of salty tidal water into urban water supplies.

Table 3.3 The seven major dams in the Hunter catchment, listed in descending order according to storage capacity (Source: NSW Department of Water (Online)).

Major Water Reservoirs	Location	Volume (ML)	Height (m)
Glenbawn Dam	Hunter River	870 000	100
Glennies Creek Dam	Glennies Creek	283 000	67
Lake Liddell	Bayswater Creek	148 000	-
Grahamstown Reservoir	-	132 000	-
Plashett Reservoir	Saltwater Creek	65 000	-
Chichester Dam	Chichester River	22 000	43
Lostock Dam	Paterson River	20 000	38

Construction of the various water retaining structures has altered the natural flow regime of rivers in the Hunter catchment. Glenbawn Dam has reduced mean annual flows in the Hunter River (Figure 3.4) along with a reduction in the size and frequency of intermediate and high flows. The valve capacity of the dam limits peak flows to a maximum of 8 x 10^3 ML/day, equivalent to a small flood with a natural return period of 1.15 years, as the dam is not being overtopped (Erskine, 1985). Furthermore, regular releases have increased flows during naturally low flow periods. Similarly, Lostock and Chichester Dams have also reduced natural low flow periods; however, frequency of large floods was not altered as water spills freely over the dam wall during high flows (Rolls *et al.*, 2011). Such modifications to the flow regime have impacted local fish communities (Gehrke and Harris, 2001), reduced the downstream transfer of sediments and altered stream morphology (Erskine, 1985). Furthermore, these structures can also be barriers to in-stream fauna. Even though Seaham Weir is fitted with a submerged orifice fishway (NSW DPI, 2006a), it remains impassable to local species due to unsuitable design (Rolls, 2011). Despite the seven large water reservoirs in the catchment, the system is not highly regulated with only 15 percent of flows being intercepted (Chessman *et al.*, 1997).

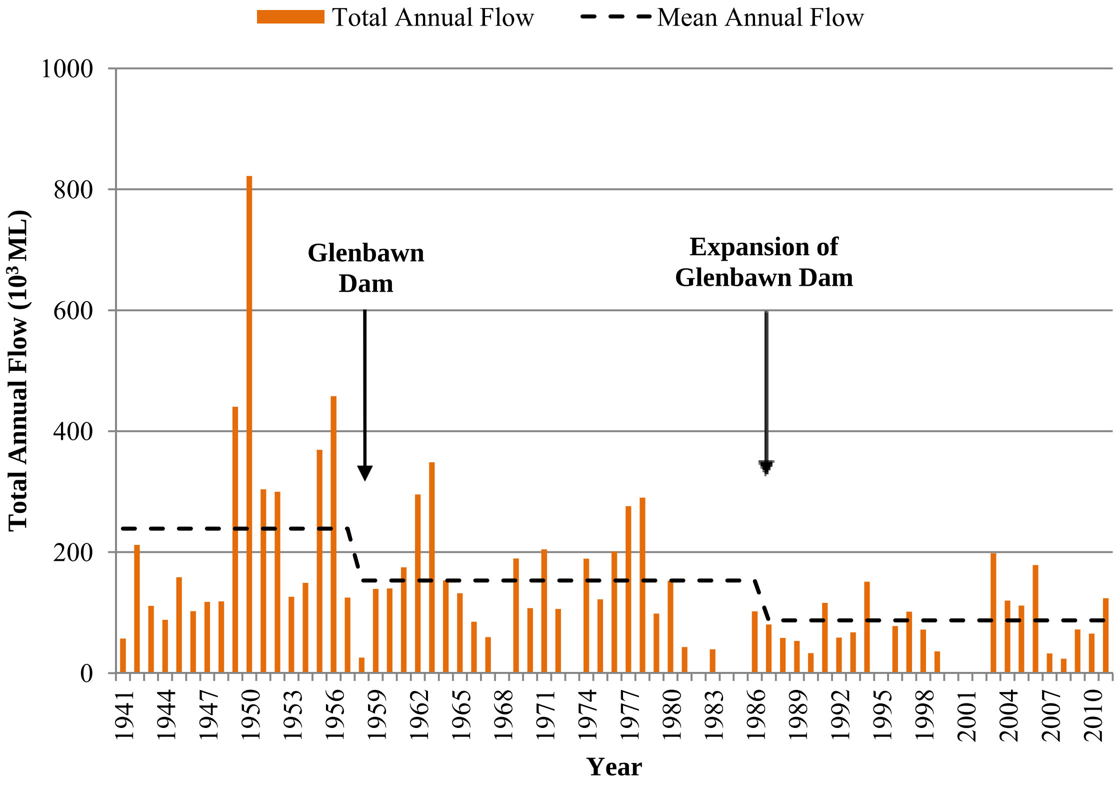


Figure 3.4 Long-term total annual flows on the Hunter River downstream of Glenbawn Dam (1941-2011, excluding 1968, 73, 82, 84, 85, 95 and 2000 to 2002) (Source: NSW Department of Water (Online), Gauging Station 210015).

3.5 VEGETATION

There is a large diversity of vegetation communities within the Hunter catchment (Peake, 2003). The dominant vegetation types are grasses, eucalypts and rainforests (Story, 1963). Open grassy woodlands were characteristic of the Merriwa Plateau (Figure 3.2), where predominantly Eucalypt (*E.*) woodlands of white box (*E. albens*), yellow box (*E. melliodora*) and red gum (*E. blakelyi*) were scattered within grasslands of *Austrostipa aristiglumis* (Peake 2003). Grasses also occur as minor communities throughout the rest of the catchment (Story, 1963). Other plant species include *Themeda australis* and *Danthonia spp.*, which were present across the catchment, and *Poa spp.* in the high regions of the Mt Royal and Liverpool Ranges (Story, 1963).

A number of different Eucalypt dominant forests and woodlands occur throughout the Hunter catchment. Wet sclerophyll forest and subalpine woodlands are found in the northern mountain ranges, where snow gum (*E. pauciflora*), white gum (*E. dalrympleana*) and silver-top stringybark (*E. laevopinea*) are common tree species (Peake 2003, Story 1963). The southern mountains support both wet and dry sclerophyll forests, with smooth bark apple (*Angophora costata*) abundant in the latter. The central Hunter region surrounded by the mountains was characterised by numerous gums, boxes, stringybarks and ironbarks, with the southern region supporting an understory of sclerophyll shrubs, including *Proteaceae*, *Leguminosae* and *Epacridaceae* (Story, 1963).

Three types of old-growth rainforests are located in the steep and high rainfall region of the Barrington Tops (Figure 3.2). These include subtropical, warm temperate and cool temperate rainforest types. A variety of plants including palms, figs, buttress roots, ferns and epiphytes are found in the subtropical rainforest (NSW DECC, 2010), while the warm temperate rainforest type is characterised by sassafras (*Doryphora sassafras*) (Story, 1963). Antarctic beech (*Nothofagus moorei*) is dominant in the cool temperate rainforest, with soft tree ferns (*Dicksonia Antarctica*) making up the understorey (Story, 1963).

Open woodlands and grasslands dominated the alluvial floodplains of the Hunter River (Peake, 2003). Cabbage gum (*E. amplifolia ssp.*) and forest red gum (*E. tereticornis*) were common floodplain trees in the lower Hunter. Rainforest covered the banks of the lower

Paterson and Williams Rivers, where red cedar (*Toona ciliata*) was abundant. Open woodlands of river red gum (*E. camaldulensis*), forest red gum and yellow box (*E. melliodora*), interspersed with kangaroo (*Themeda australis*) and wallaby (*Danthonia spp.*) grass, were common in the floodplains of the mid and upper Hunter (Peake, 2003). The riparian zones of the region were once heavily vegetated, although widespread clearing of native vegetation has occurred with only 0.7 percent of the valley floor under conservation (Peake, 2003). Currently, remnants are present of river oak (*Casuarina cunninghamiana*) and swamp oak (*Casuarina glauca*) along waterways, with other native riparian vegetation occurring sparsely (Hoyle *et al.*, 2008; Chalmers *et al.*, 2012). Undesirable species such as willows, herbaceous weeds and exotic grasses dominate the riparian margin (Hoyle *et al.*, 2008; Kyle and Leishman, 2009).

The intertidal region of the Hunter River estuary contains mangroves and various saltmarsh species. The second largest area of mangroves in NSW is located in the Hunter River estuary along the water channels and on mudflats, with gray mangrove (*Avicennia marina*) and river mangrove (*Aegiceras corniculatum*) being the most common species (Peake 2003). Saltmarshes are found away from the water adjacent to the mangroves, still within the tidal reaches. Common species include *Sarcocornia quinqueflora*, *Suaeda australis*, *Sporobolus virginicus* and *Triglochin striatum* (Peake 2003). Remains of a rainforest occur in the Kooragang wetlands, supporting ash tree and red cedar. *Melaleuca spp.* is a characteristic tree commonly occurring in Hexham swamp (Story, 1963). Wetlands have been severely disrupted through vegetation clearing and restriction of tidal flows, for which floodgates are commonly used. A number of conservation areas where this vegetation is protected occur in the Hunter River estuary, including Kooragang Wetland, Kooragang and Hexham Nature Reserve.

3.6 LAND USE

Agriculture is the dominant land use of the Hunter catchment, although there are other significant land uses. The first farm was established on the lower Hunter River floodplain in 1812 (Perry, 1965). Since then, substantial amounts of the mid-Hunter catchment have been cleared for agricultural purposes (Peake, 2003). Livestock grazing for beef, sheep, wool and dairy production is the dominant agricultural activity in the catchment. Although, cropping, poultry and egg production also make up substantial industries (Chessman *et al.*, 1997). Furthermore, the central lowlands are well known for supporting some of the oldest wineries in Australia. Large underground and open-cut coal mines have developed in the central lowlands of the Hunter catchment around the towns of Newcastle, Singleton and Muswellbrook. In 1985, approximately 65 mines were in operation in the Hunter region (Day, 1988), while between 2006-2007 the number of operating mines had dropped to 34 (The Hunter Valley Reserach Foundation, 2009). This is due to the geological feature known as the Sydney Basin, which is well endowed with deposits of Permian coal (Day, 1988). Two thirds of the coal of New South Wales is extracted in the region (Chessman *et al.*, 1997), making it the most significant export in the Hunter catchment with a value of $900 million (1982-83) (Day, 1988). Coal is also used locally in nearby industries for electricity generation or steel production (Chessman *et al.*, 1997). The region is one of the most developed in NSW with a population of approximately half a million people (Day, 1988). Most of those people are located in the Newcastle metropolitan area, which is the second largest urban area in NSW after Sydney. Other larger urban centres in the Hunter catchment include Cessnock, Maitland, Singleton and Muswellbrook.

The Hunter catchment and its river system have been subject to widespread anthropogenic impacts since European settlement. Originally, more than 90 percent of the mid-Hunter catchment supported native forests and woodlands (Peake, 2003). Much of the region has been cleared for agricultural purposes, with only 20 percent of the land area currently under formal conservation (Peake, 2003). The majority of protected areas are located in the inhospitable rugged northern and southern mountainous parts of the catchment, where some of the larger national parks in the region include the Barrington Tops, Goulburn and Yengo National Parks. In addition to the mass clearing of floodplains, the river systems of the Hunter have been impacted by other anthropogenic activities. These have included changes in the flow regime due to water resource development (refer to Section 3.4), removal of in-stream sediments (Erskine *et al.*, 1985), river training works for increased channel stability (Erskine, 1992), flood mitigation works such as channel straightening, construction of artificial cutoffs, levees, removal of bank vegetation and large woody debris (Erskine and Webb, 2003). Development of extensive infrastructure networks (roads, dams, etc.) (Williams and Watford, 1997) to support the coastal populations and industry are affecting native ecosystems.

3.7 AQUATIC BIOTA

Australia is home to more than 200 freshwater fish species (Allen *et al.*, 2002). Of these species, 86 endemic and 22 introduced species are located on the south eastern coast (McDowall, 1996). Seven exotic and 52 native freshwater and estuarine species live in the Hunter catchment and the near surroundings (NSW DPI, 2006b). Distinct fish communities occur in the montane streams while species richness is higher downstream (Gehrke and Harris, 2000). Majority of the endemic species are still commonly found in the region, with the exceptions of black cod (*Epinephelus daemelii*) and green sawfish (*Pristis zijsron*) which are both on the endangered list. The bullrout (*Notesthes robusta*) and freshwater herring (*Potamalosa richmondia*) have a limited, yet not threatened abundance, and the estuary perch (*Macquaria colonorum*) and Australian bass (*Macquaria novemaculeata*) have an uncertain abundance (NSW DPI, 2006b). Goldfish (*Carassius auratus*), gambusia (*Gambusia holbrooki*) and the common carp (*Cyprinus carpio*) are widespread exotic species in the northern coastal catchments (Gehrke and Harris, 2000), occurring in still or gently flowing waters (McDowall, 1996). Exotic species degrade habitats and exhaust food supplies of endemic species.

The numerous fish species in the Hunter catchment possess different migration patterns, moving between various aquatic habitats predominantly for breeding purposes. A list of all the species is provided in Appendix A. Three of these species migrate only within freshwater (potamodromous pattern), while 21 carry out migrations between freshwater and sea water (NSW DPI, 2006b). These include 13 amphidromous fish (migrations between the sea and freshwater but not for breeding), one anadromous fish (migrating from the ocean to freshwater for breeding) and seven catadromous fish (migrating from freshwater to the ocean for breeding). The migration pattern of the other species is either localised or unknown. Australian bass is an iconic catadromous freshwater fish on the Australian south-eastern coast and a popular angling species. Females migrate from freshwater to estuaries for spawning in winter, with juveniles migrating back upstream in spring and summer (Harris and Rowland, 1996). Males generally remain in estuarine or lowland habitats, while females migrate upstream back into freshwater (Harris, 1987). Female Australian bass are found in lagoons and upstream lotic habitats commonly associated with boulder and gravel pools (Harris, 1987; Koehn and O'Connor, 1990).

The waters of the Hunter catchment provide habitat to numerous aquatic biota, other than just fish species. Six species of freshwater mussels have been recorded in the catchment. *Hyridella drapeta* is the most abundant species, *Hybridella depressa* is only found in a small community on the Williams River, while *Alathyria profuga* is the most widespread species occurring throughout various sub-catchments (Jones and Byrne, 2010). School prawns (*Metapenaeus macleayi*) are an economically valuable species that is found in the estuary of the Hunter River (Ruello, 1973). School prawns spawn in the ocean and juveniles migrate upstream into estuaries, until they are ready to spawn and migrate back to the ocean. They are commonly trawled in the estuary and used either as bait or for human consumption. The estuarine waters and associated wetlands host numerous other biota, including crustaceans, worms, frogs and birds (Hodda and Nicholas, 1985).

Chapter 4

4 Chapter 4 – Anthropogenic Change in Longitudinal Connectivity of the Hunter River

4.1 INTRODUCTION

Connectivity is a central component of the structure and functioning of riverine landscapes. Freshwater aquatic ecosystems are dependent on hydrological connections, as they construct and link freshwater habitats, in addition to mediating other essential processes. The spatial organisation of physical components of riverine landscapes and their respective connections influence patterns and processes of river ecosystems (Power and Dietrich, 2002; Benda *et al.*, 2004; Campbell Grant *et al.*, 2007). The frequency of anthropogenic barriers have significantly altered and degraded the connectivity of stream networks in Australia (Harris, 1984; Arthington and Pusey, 2003) and global (Nilsson *et al.*, 2005) river systems, thereby posing a major threat to the health and persistence of these ecosystems. Understanding the character and extent of anthropogenically induced changes is an important first step towards informed management of river ecosystems.

This chapter investigates the extent to which stream network connectivity of the Hunter River has changed as a result of anthropogenic fragmentation. It explores the influence multiple types of barriers have on the connectivity and character of the stream network, especially as studies to date have predominantly focused on fragmentation by a single barrier type without acknowledging the presence and cumulative effects of multiple barrier types on connectivity (e.g. Anderson *et al.*, 2008; Park *et al.*, 2008; Walter and Merritts, 2008). In addition, this research also considers whether certain regions of the stream network are at greater risk of change and degradation by fragmentation than others. This chapter provides the foundation for the artificially fragmented stream network which is utilised in the development of an optimisation restoration model in Chapter 5, and guiding decisions on Hunter River connectivity restoration in Chapter 6.

4.2 METHODS

4.2.1 Introduction

The methods employed to assess the change in connectivity of the Hunter River network are made up of three main components and these are described below. The first section describes the methods used to characterise the natural state of connectivity of the Hunter River before anthropogenic disturbance. Two opposing river system theories are applied in the characterisation, with the river network as both a physical continua (Vannote *et al.*, 1980) and a hierarchical patch river system (Poole, 2002; Thorp *et al.*, 2008). Section two describes the methods used to assess anthropogenic fragmentation of the Hunter River network, in terms of the initial barrier identification process and secondary network fragmentation. In the third section, the methods of data analysis are outlined. All spatial characterisation and analysis in this chapter was carried out in *ArcGIS Ver. 9.3*, with the use of *Hawth's Tools Analysis Toolpack* and *X-Tools Pro* add-ons. Statistical analyses of the data were undertaken using *PATN* and *Primer 6* software.

4.2.2 Natural Connectivity

4.2.2.1 River Types: Stream Network as a Patch Mosaic

The first technique used to determine the natural stream connectivity of the Hunter River catchment employs a hierarchical theory approach. The stream network was classified into patches, referred to as 'River Types' (Harris *et al.*, 2009), based on a statistical classification using an array of geomorphic variables. Fifteen geomorphological variables representing regional, valley and channel scales were used to classify stream character. The methods employed to classify the physical template of river channels within the Hunter stream network are outlined in the following three sections. The first section discusses the source data used in the variable extraction process. The second section describes the method employed to extract the necessary geomorphic variables. The third section outlines the process involved in defining the River Types from the geomorphic variables. The methods used in this thesis are similar to those applied by Thoms *et al.* (2007), Harris *et al.* (2008) and Scown (2010).

4.2.2.1.1 Source Data

Four regional spatial data sources were utilised to calculate the fifteen variables used in the identification of River Types within the Hunter River catchment: a digital elevation model (DEM), and digital spatial data on geology, streamlines and rainfall. The DEM was obtained from Geoscience Australia (Datum: GDA 94) and constructed from a 1:100 000 topographic map, with each pixel having a 20.4 x 20.4m ground resolution. A spatial digital dataset of the geology of the region was also acquired from Geoscience Australia and this dataset is composed of polygons associated with five lithological groups of the catchment, namely: igneous felsic, igneous mafic, organic-rich sediments, sedimentary rocks and regolith. Digital streamlines of the named watercourses in the Hunter catchment were acquired at a 100 000-scale. Streamlines were cross-checked with recent Landsat images to validate the quality of the stream network. Streams shorter than 5km were removed as they were at a finer scale than the variable sampling; sampling points occurred at 5km spacing. Rainfall data were obtained in the form of a digital rainfall raster grid from the Australian Government Bureau of Meteorology with rainfall data collection occurring between 1961 and 1990. The raster grid provides mean annual rainfalls of the catchment, at a 2.5 x 2.5km on-ground resolution.

4.2.2.1.2 Extraction of River Type Variables

The River Typing process involved the extraction of fifteen geomorphic variables at each sample site along the stream network. Sample sites were automatically created on the stream network of the Hunter River at 5km intervals in an upstream direction. Variables were extracted at a regional, valley and channel scale (Table 4.1). Regional variables are comprised of the rainfall, geology and elevation at the sample site. Valley width, valley trough width, width to trough width ratio, both left and right valley slopes, and finally the longitudinal valley slope made up the valley scale variables extracted at the sample site. Lastly, channel character at the sample site are represented by the channel’s planform, number of channels, channel sinuosity, channel belt width, channel belt wavelength and channel belt sinuosity.

All three regional scale variables, rainfall, geology and elevation, were determined with surface spot tools in *ArcGIS 9.3*, where values underlying the sample sites were extracted from the rainfall raster grid, geological polygons and DEM respectively.

Table 4.1 Descriptions of the fifteen geomorphic variables used in the River Typing process and how they were calculated at each sample site, after Thoms *et al.* (2007).

Scale	Variable	Abbreviation	Description	Calculation
Region	Rainfall	R	Mean annual rainfall at sample site	Extracted from rainfall grid
	Geology	G	Underlying geology class at sample site	Extracted from geology polygons
	Elevation	H	Elevation above mean sea level at sample site	Extracted from DEM
Valley	Valley width	Vw	Distance between the highest points of the valley on each side of the streamline at sample site	Figure 4.1
	Valley trough width	Vtw	Width of the valley bottom where the slope is less than three percent at sample site	Figure 4.1
	The ratio Vw:Vtw	Vw:Vtw	The ratio of valley width to valley trough width at sample site	$\frac{Vw}{Vtw}$
	Valley slope left	VsL	Average slope from channel to valley summit on the left side at sample site	$\frac{Left\ summit\ elevation - H}{Distance\ from\ site\ to\ summit}$
	Valley slope right	VsR	Average slope from channel to valley summit on the right side at sample site	$\frac{Right\ summit\ elevation - H}{Distance\ from\ site\ to\ summit}$
	Longitudinal valley slope	Vs	Average slope of stream segment between sample site and the next site downstream	$\frac{Elevation\ loss}{Channel\ segment\ length}$
Channel	Planform	P	Whether the channel segment downstream of sample site is meandering (single channel) or anabranching (multiple channels)	Visually identified
	Number of channels	NoChan	Number of channels occurring in the segment downstream of sample site	Visually identified
	Channel sinuosity	Csin	Sinuosity of the main channel segment downstream of sample site	$\frac{Channel\ length}{Straight\ line\ distance}$
	Channel belt width	CbW	Width of the channel meander belt at sample site	Figure 4.2
	Channel belt wavelength	CbWave	Average wavelength of both sides of the channel belt for the entire streamline on which a sample site is located	Figure 4.2
	Channel belt sinuosity	CbSin	Sinuosity of the left side channel belt for the segment downstream of sample site	$\frac{Left\ channel\ belt\ lengh}{Straight\ line\ distance}$

The DEM of the Hunter River catchment was utilised to calculate all six valley scale variables. Longitudinal valley slope at a sample site was measured between respective sample sites and the following downstream site, with the elevation and channel length data. Automated tools were used to calculate the first five variables from elevation-transect data obtained at each sample point (Table 4.1). Transects extending from channel to adjacent valleys were automatically created at each sample point, perpendicular to the respective streamline (Figure 4.1). Elevation-transect data were obtained by creating equidistant points along the length of the transect, and extracting elevation data of points with surface spot tools from the underlying DEM.

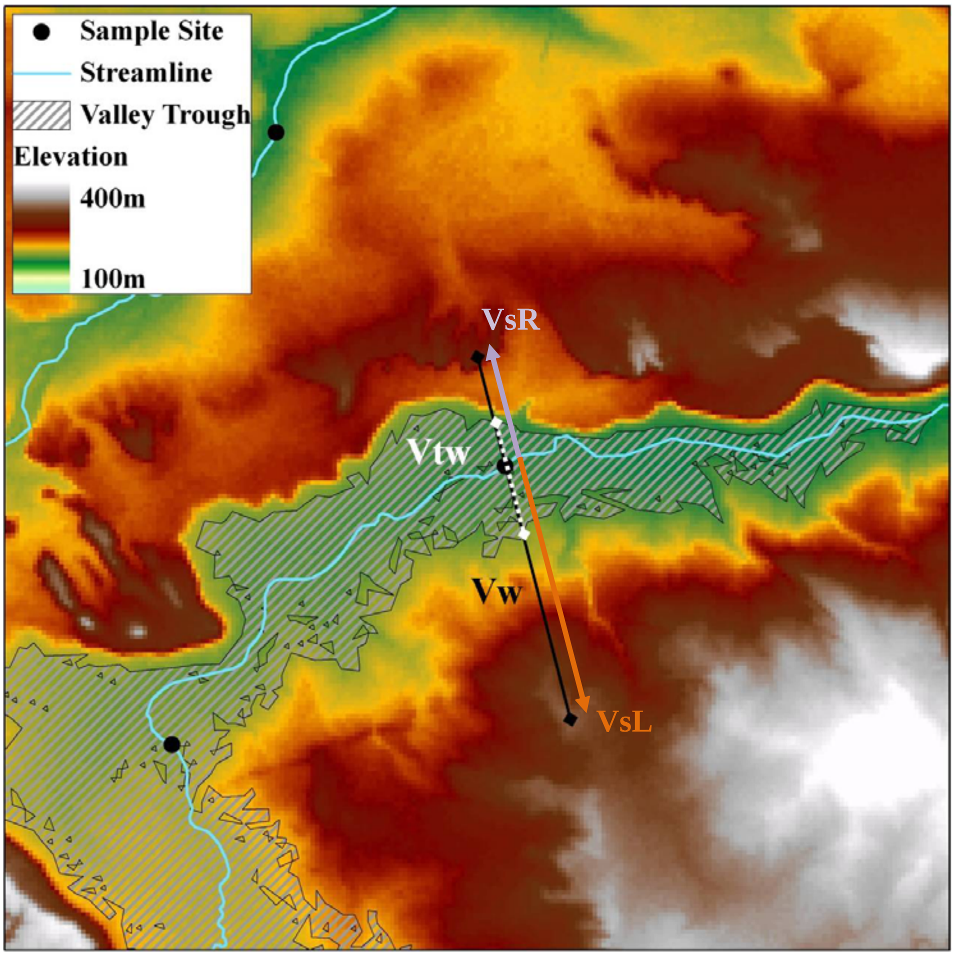


Figure 4.1 Calculation of valley width (Vw), valley trough width (Vtw), left and right valley slopes (VsL and VsR) at sample site. All variables calculated from DEM (Source: Scown, 2010).

Channel planform and the number of channels in each segment downstream of a sample site were determined visually. Channel sinuosity of the stream segment extending downstream of each sample site was measured with *Hawth's Tools* add-on. Channel belts were delineated manually for calculation of the three channel belt variables. The X *Tools Pro* add-on was used to create points on both channel belt lines adjacent to the sample site. Channel belt width was calculated as the distance between points on channel belts adjacent to each sample site, while channel belt sinuosity and channel belt wavelength were calculated on channel belt segments downstream of points (Figure 4.2; Table 4.1). All channel belt variables were calculated using the *Hawth's Tools* add-on.

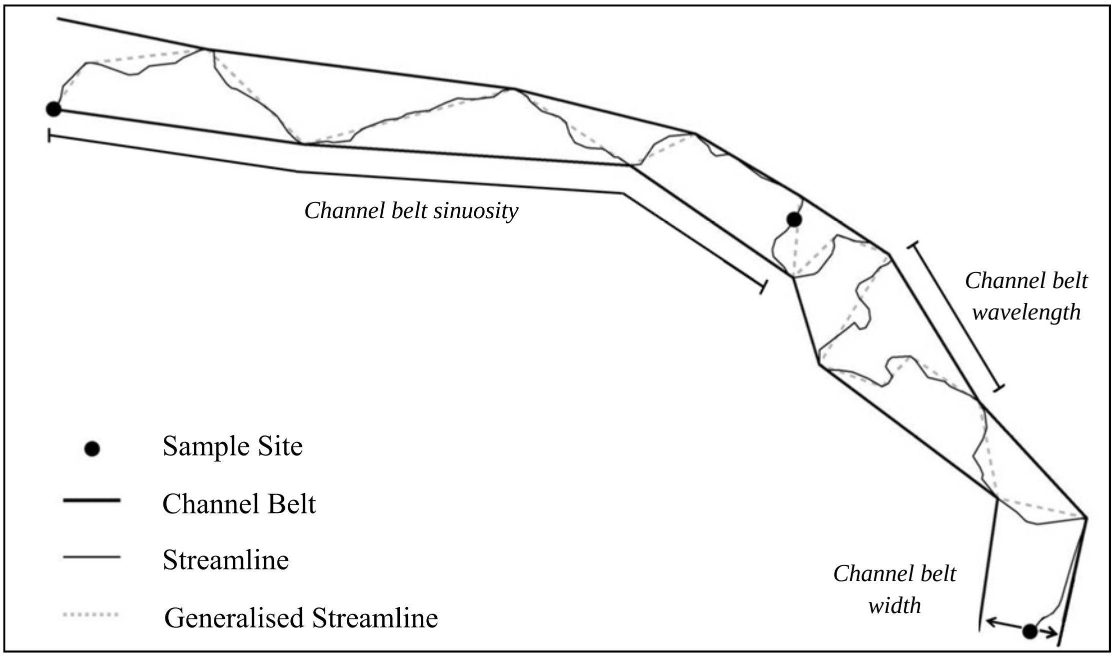


Figure 4.2 Calculation of the three channel belt variables at sample sites, from manually delineated channel belts of meandering streamlines (Source: Scown, 2010).

4.2.2.1.3 River Type Definition

A series of multivariate statistical analyses were used to determine and classify sample sites in the Hunter River, so as to group sample sites with similar physical characteristics based on the fifteen geomorphic variables. Initially, a hierarchical non-congomarative statistical clustering analysis was undertaken within the *PATN* software package, where the optimum number of self-emerged groups with the greatest level of within-group association was determined. Each sample site was then associated with each self-emerged group; also referred to as a 'River Type'. The entire 5km segment downstream of the sample site was assumed to have the same physical characteristics as the sample site, and thus was of the same River Type. At all sample sites where two segments of the same River Type met, the segments were merged into a single continuous segment of the same

River Type, while meetings of segments of differing River Types remained untouched. River Type groups were named based on their location in the stream network and the dominant geomorphic variable in the formation of the group, which was calculated with a similarity percentage analysis (SIMPER). The stream network characterised in relation to the underlying River Types was referred to as the River Character template.

The traditional concept of depicting river systems as a longitudinal continua (Vannote *et al.*, 1980) was also applied to determine the natural state of connectivity of the Hunter stream network, and this was referred to as the Continuum template. The stream network was characterised as a continuous unit, with the only discontinuities in the network occurring at natural physical barriers, i.e. waterfalls, which defined the extents of stream segment boundaries. The set of streamlines used in the analysis were the same as for the River Typing, to ensure a consistency in results, although the actual River Types were not considered.

4.2.3 Anthropogenic Fragmentation

4.2.3.1 Barrier Identification and Distribution

Locations of natural barriers were obtained from the New South Wales Department of Industry and Investment and these were imported into *ArcGIS 9.3* as a point shapefile. The river network streamlines were split at the points where natural barriers occurred, with the use of the *X-Tools Pro* add-on tool, and river segment lengths extracted.

In this study, four types of anthropogenic barriers were identified to fragment the Hunter River network, namely: floodgates, dams, weirs and road crossings. These identified structures are widely accepted to act as either physical or hydraulic barriers to different components of the riverine system (refer to Chapter 2, Section 3.3). Combination of various source data (Table 4.2) and methods were used to identify each barrier type.

Floodgates and weirs were wholly identified through barrier information supplied by NSW Industry and Investment and NSW Water Information. An inventory of barriers of different types and their physical locations was obtained from NSW Industry and Investment and NSW Water Information. Barrier data were divided according to barrier type, GPS coordinates were imported into *ArcGIS 9.3* for each type individually and overlayed onto the existing river network. Point shape files of each barrier type were produced; however, barriers that did not fall onto the streamlines as defined in this study were removed and not considered in the investigation. Dams in the river network were identified with the use of spatial imagery taken in January 2011; these being the most recent images available at the time of data collection. Landsat TM (USGS) images were viewed in pseudo-natural colour composite (Red-Green-Blue – 541) to accentuate visibility of water bodies. The dams were identified by locating blue 'patches' which fell upon the defined river network, although no ground-truthing was performed to validate the existence of what were identified as smaller dams.

Digital layers of road and rail networks, obtained from Geoscience Australia, were intersected respectively with the defined river network in order to locate the road crossings. A point shape file of the intersections was made with the use of an *ArcGIS 9.3* add-on (*Hawth's Tools*). All points of intersection were cross-checked with Google Maps satellite imagery, because of the high definition imagery and up-to-date infrastructure data, and adjusted as necessary. In addition, data obtained from NSW Industry and Investment were further used to confirm the road crossing dataset. The road crossing dataset was composed of numerous structures, including culverts, causeways, fords and bridges, which are all known to be barriers in certain circumstances (Warren and Pardew, 1998; NSW DPI, 2006b; Nislow *et al.*, 2011). No differentiation was made between the various types of road crossings as adequate data were not available. As a consequence, all road crossings have been conservatively classified as complete barriers, even though this might not always be the case. Additional research and fieldwork is required to identify and differentiate the actual barriers from the potential barrier dataset compiled in this thesis.

Table 4.2 List of data sources used in identification of each barrier type.

Barrier Type	Data Sources
Floodgate	NSW Industry and Investment
Dam	Landsat TM (January 2011)
Weir	NSW Industry and Investment and NSW Water Information
Road Crossing	NSW Industry and Investment, Geoscience Australia (Road/Rail network GIS Layers) and Google Maps 2011

The spatial distribution of barriers in relation to the various River Types was assessed. Each barrier type was manually identified in *ArcGIS 9.3* in relation to the respective underlying River Type. The data were divided into groups according to both barrier type and River Type, thereby providing information on the barrier distribution within each River Type of the Hunter River network.

4.2.3.2 Network Fragmentation

Two natural states of connectivity - the River Character and Continuum river network templates - were used to investigate the change in connectivity of the Hunter River in relation to barrier types. The state of fragmentation of each natural connectivity template was assessed five times, first exploring fragmentation associated with floodgates, dams, weirs and road crossings individually, and then all barrier types. Numerous copies were made of both sets of natural streamlines, so each could be fragmented by the different barrier types independently. Barrier type point shape files were overlayed onto the streamlines in *ArcGIS 9.3*, and streamlines were split with *X-Tools Pro* add-on, at all points where barriers overlapped streamlines. Streamline segments were considered as continuous entities, edges only occurred at points where a barrier was located, or where two differing River Types met if the River Character template was used (Figure 4.3). Methods were repeated for both templates and all individual barrier types.

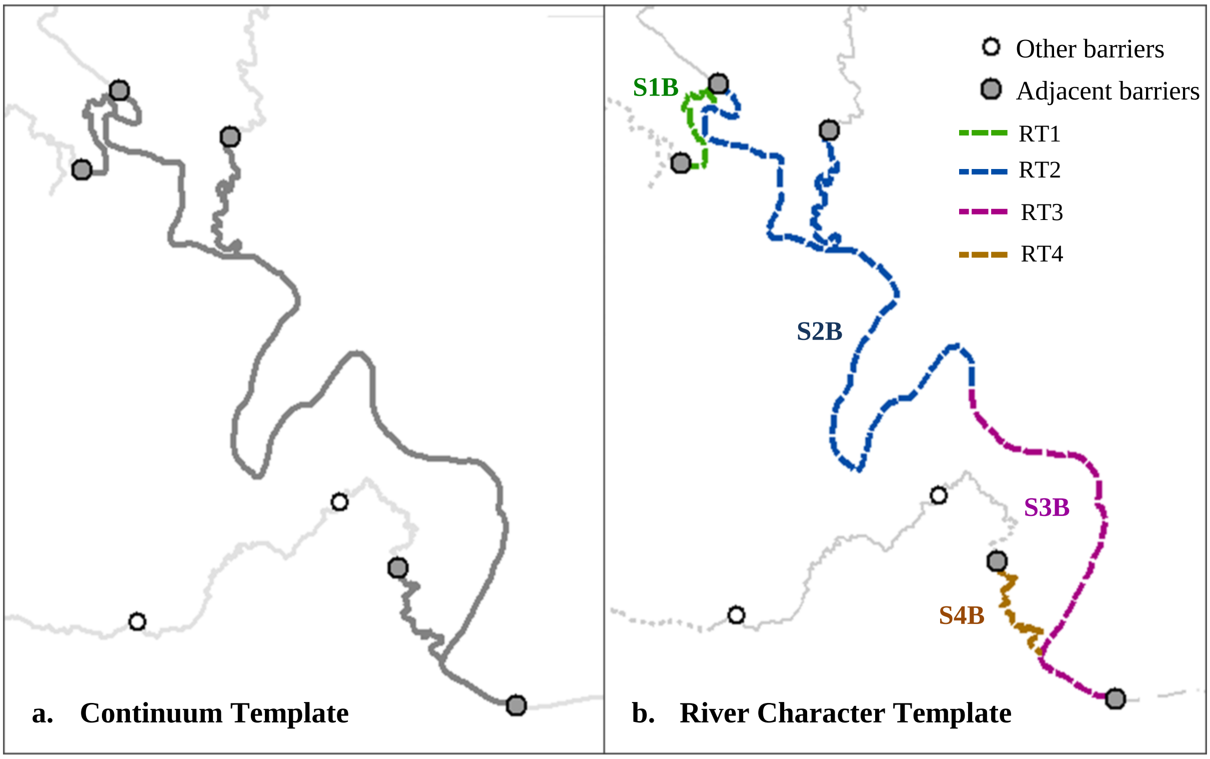


Figure 4.3 Example of a). A single streamline segment for Continuum template. b). Same segment further divided into four segments (S1B, S2B, S3B and S4B) according to underlying River Types.

4.2.4 Data Analysis

4.2.4.1 Barriers

Analysis of the character of barriers in the Hunter River consisted of an abundance analysis of barrier types and barrier distribution. Abundance of barrier types was obtained through basic proportion calculations. The distribution of barriers in relation to each River Type was assessed firstly by basic abundance calculations, i.e. looking at total number of barriers per barrier type per River Type. Secondly, the 'evenness' of distribution of barriers throughout the various River Types was examined with a newly developed 'Distribution Index' (DI) (Equation 4.1). The Distribution Index is a measure which scales barrier abundance within a River Type in relation to that River Type's stream network abundance, allowing for barrier distribution comparisons within different parts of a network to be made.

$$DI_i = \frac{pB_i}{pL_i}$$

Equation 4.1 Barrier Distribution Index

Where: DI_i = Distribution Index for the i^{th} River Type

pB_i = proportion of total number of barriers located in i^{th} River Type

pL_i = proportion of total stream length comprised of i^{th} River Type

A DI value of less than one indicates a lower-than-expected number of barriers in a River Type (lower fragmentation), a DI value of one indicates an even barrier distribution in a specific River Type, and a DI value of greater than one signifies excessive number of barriers in the specific River Type.

4.2.4.2 Stream Network

The natural and fragmented stream network datasets were analysed with a number of tools. Firstly, methods used to perform segment length and segment abundance analyses on all the stream networks were defined. Datasets obtained from the River Character network were further analysed and compared with multivariate statistical tools. Then, a Dendritic Connectivity Index (Cote *et al.*, 2009) was used to quantify the extent of the Continuum network. The methods employed to perform these analyses are outlined below in greater detail.

4.2.4.2.1 Segment Length and Abundance Analysis

Fragmentation of the various stream networks was analysed with the streamlines' respective segment length and segment abundance data. Segment length data for each stream network was obtained from fragmented streamlines in *ArcGIS 9.3*. A length attribute was added to each streamline shape file with *Hawth's Tools* add-on. Streamlines' attribute tables were exported, which included length and River Type attributes for each segment within the streamline set. Segment data for each fragmented stream network was grouped into length range categories according to River Types, from which segment length distribution and segment number distribution histograms were produced for each River Type and the entire stream network. Additionally, mean segment lengths, proportion of entire stream network and proportion of each individual River Type made up by the longest segment respective were calculated for all stream networks. Stream network datasets were handled in the same manner irrespective of which template the dataset originated from. The methods previously outlined were in reference to stream networks originating from the River Character template. However, Continuum template datasets were analysed without segment data being divided according to the River Type.

4.2.4.2.2 Multivariate Statistics

A series of multivariate statistical tools were employed to analyse and compare the various stream network data, originating from the River Character template. Segment length data for each of the fragmented networks, statistical groups, were divided according to segment River Type, within group individuals. Eight variables were calculated from segment length datasets for each within group individual, as an accurate representation of the fragmented character of the individual, namely: number of segments, mean segment length, median segment length, maximum segment length, minimum segment length, standard deviation, and 25th and 75th percentiles. The natural stream network, the floodgate-fragmented network, road crossing-fragmented network, dam-and-weir fragmented network and stream network fragmented by all barriers were statistically tested. Between groups similarity was determined with an analysis of similarity (ANOSIM), while within group dispersion among individuals was quantified with the multivariate dispersion index. Data were then graphically represented with a two-dimensional ordination plot. Statistical tests were carried out with *PATN* and *Primer 6* software package.

4.2.4.2.3 Dendritic Connectivity Index

The degree of connectivity of the Continuum river network template of the Hunter Basin was measured and quantified with the DCI: Dendritic Connectivity Index (Cote *et al.*, 2009). The DCI (Equation 4.2 to Equation 4.4) is a quantitative measure of the level of connectivity of a river system at a network scale, where both physical characteristics (segment length) and spatial layout of a river network are incorporated in calculation of the index. DCI was calculated for both the natural river network and the artificially fragmented river network, which included network fragmentation due to individual barrier types, namely: floodgates, dams, weirs and road crossings, and fragmentation due to all barrier types simultaneously. In the calculations, all barriers were assumed to be fully impassable and network segment length data were obtained from streamline shape files in *ArcGIS 9.3*. For each network, a DCI for fish of both diadromous and potamodromous migratory patterns was calculated.

$$DCI_D = \sum_{i=1}^{n} c_{ij} \frac{l_i}{L} * 100$$

Equation 4.2 Dendritic Connectivity Index: Diadromous application

$$DCI_P = \sum_{i=1}^{n} \sum_{j=1}^{n} c_{ij} \frac{l_i}{L} \frac{l_j}{L} * 100$$

Equation 4.3 Dendritic Connectivity Index: Potamodromous application

$$c_{ij} = \prod_{m=1}^{M} p_m^u \, p_m^d$$

Equation 4.4 Cumulative porosity variable used in Dendritic Connectivity Index

Where: n = total number of continuous stream segments in river network

L = total stream length of network

l_i / l_j = respective i^{th} and j^{th} segment lengths

M = total number of barriers passed between i^{th} and j^{th} segment

p_m^u / p_m^d = upstream/downstream porosity of m^{th} barrier

4.3 RESULTS

4.3.1 Introduction

There are three main sets of results; each of which are elaborated below. The first section deals with quantifying the natural state of connectivity of the Hunter catchment, with it being quantified for both the Continuum and River Character network templates. The second section describes the abundance and spatial distribution of anthropogenic barriers located in the Hunter Catchment, and the final third section details the change in connectivity of both natural states of the network as a result of anthropogenic fragmentation.

4.3.2 Natural Connectivity

The stream network of the Hunter River has a total length of 9080 km. The continuum network is dissected by five natural barriers located in the north eastern headwaters of the catchment (Figure 4.4) and as a result the longest segment has a total length of 8932 km or 98 percent of the entire stream network length. The other five segment lengths are 137 km, 6 km, 5.3 km, 1.7 km and 1.5 km respectively. The average stream segment length of the Hunter catchment as a continuum is 1500 km.

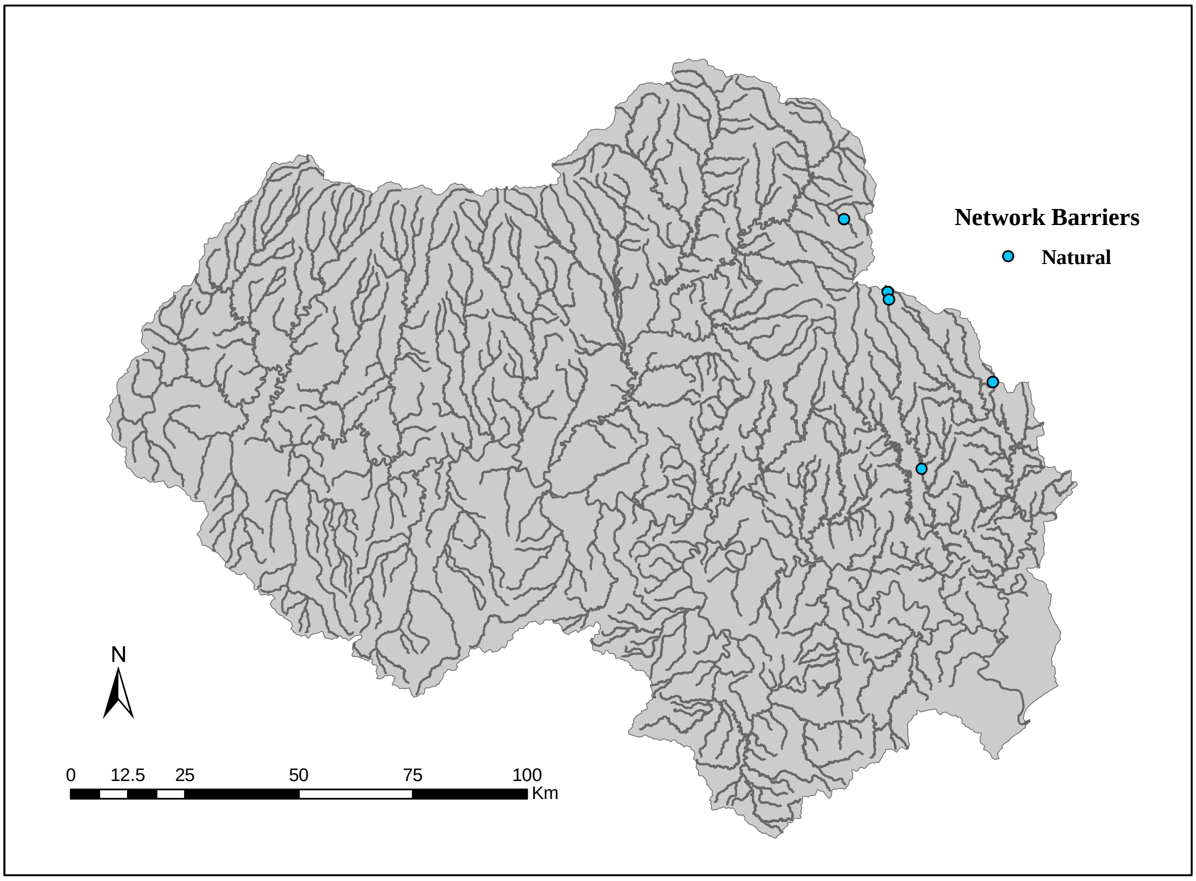


Figure 4.4 Spatial distributions of natural barriers in the Hunter River catchment.

Five distinct River Types were found to characterise the stream network of the Hunter River and their distribution resembled a patch mosaic (Figure 4.5). This network was comprised of 643 disjunct segments of the varying River Types, with an average segment length of 14 km.

The distribution of each of the five River Types throughout the Hunter catchment exhibits a distinct spatial pattern (Figure 4.5). River Type 1 was predominantly located in the headwater sections of the network. River Type 2 was relatively abundant in the Goulburn River sub catchment, with further segments sparsely spread throughout the rest of the catchment. River Types 3 and 4 were found throughout the network, although appeared to be more common in the eastern sections and associated with higher order streams in the western regions. Finally, River Type 5, the least abundant of the River Types, was present in the main stem of several of the major rivers in the catchment. In this thesis, headwater-associated River Types refers to River Types 1 and 2, while lowland-associated River Types refer to River Types 3, 4 and 5, due to the broad spatial patterns.

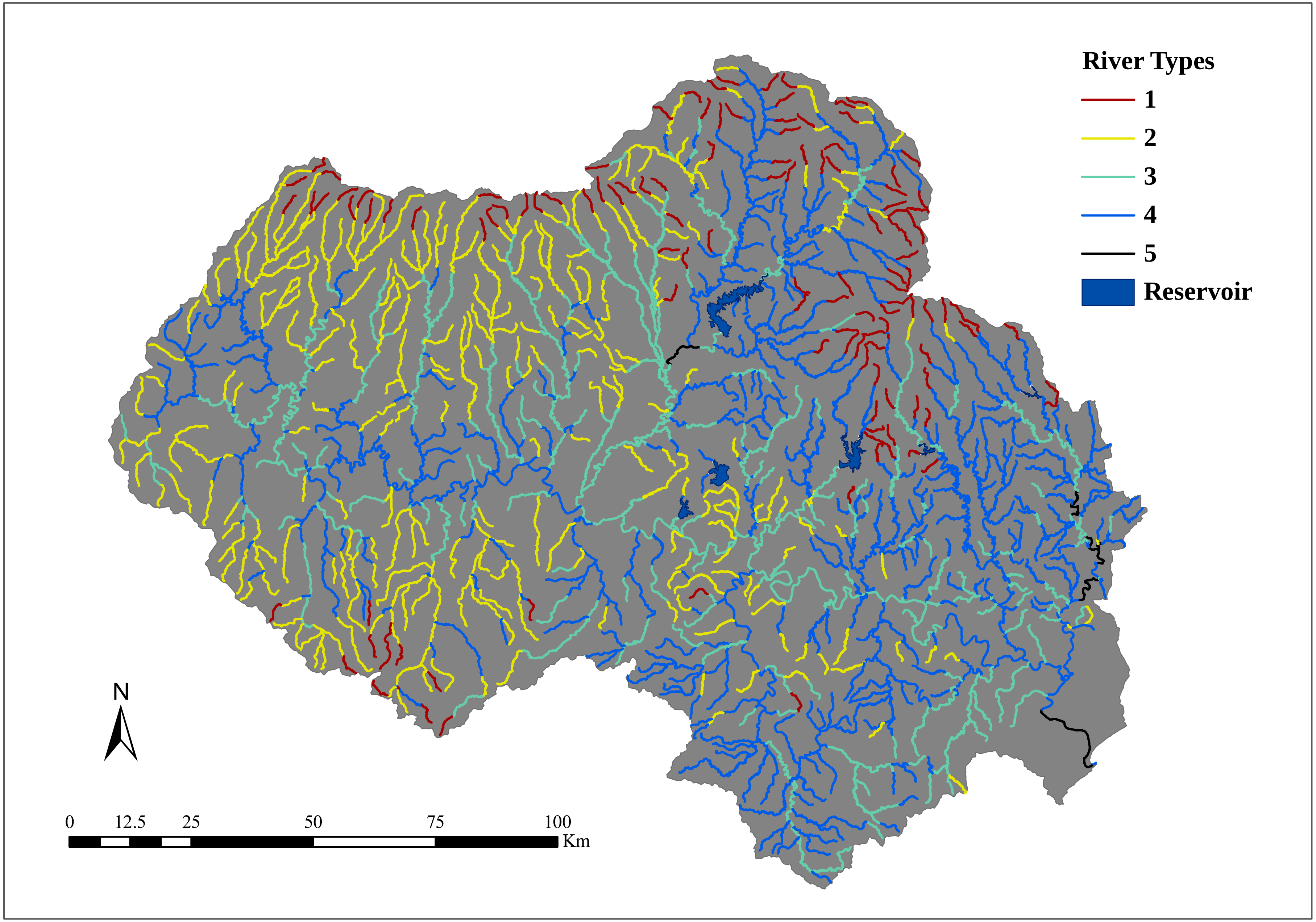


Figure 4.5 Spatial distributions of the five River Types in the Hunter River catchment.

SIMPER analysis showed that each of the fifteen geomorphic variables contributed differently to the within group similarity of the five River Types (Figure 4.6). Valley-scale variables were dominant in characterising the majority of River Types, with the exception of River Type 4, where channel-scale variables were critical. Low valley-width to valley trough-width ratio was the dominant variable (approximately 60 percent) in characterising River Type 1, associating it with constrained headwater streams (Table 4.3). Steep valley sides and wide, open valleys were attributed with River Types 2 and 3 respectively. The dominant geomorphic characteristic of River Type 4 was an above average channel sinuosity. River Type 5 was distinguished by wide valley troughs and as the only River Type with anabranching river segments. Geology appeared to be the least significant of the fifteen geomorphic variables in characterising River Types. The SIMPER analysis results were used to name each of the five River Types (Table 4.3).

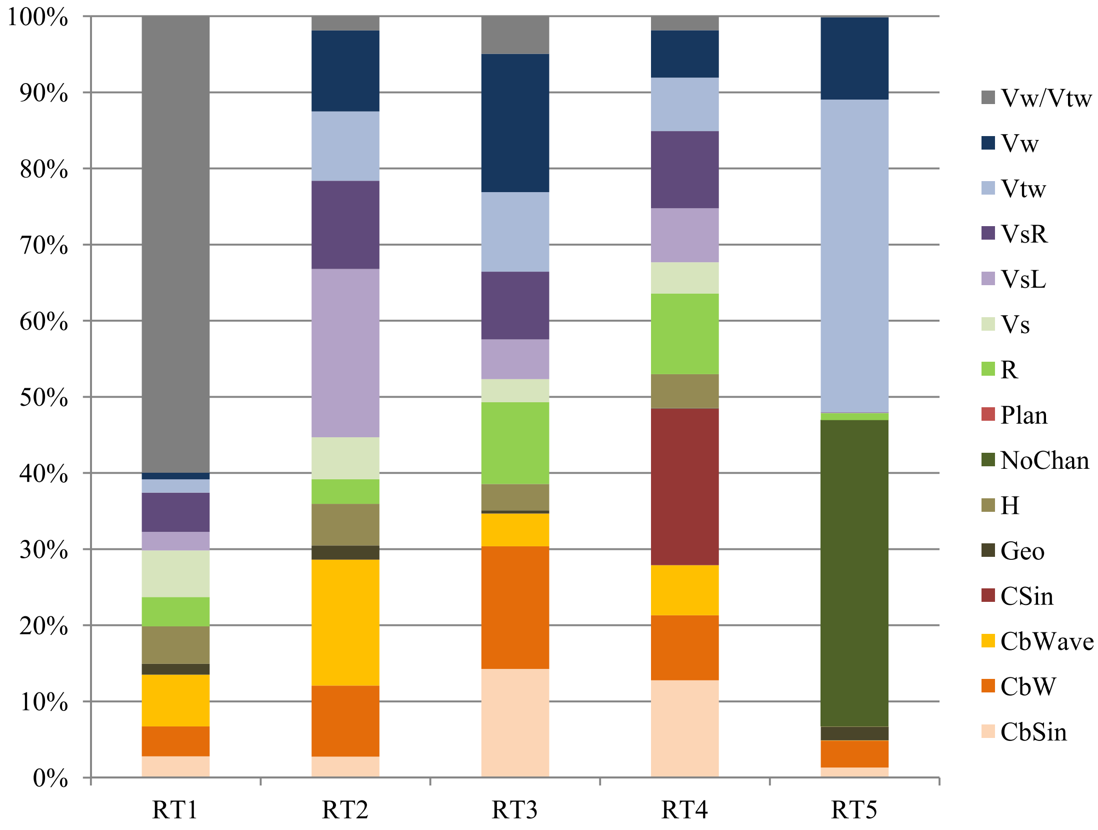


Figure 4.6 Percentage contribution of each of the fifteen geomorphic variables in determination of River Types, according to within group similarity (SIMPER analysis).

4.3.2.1.1 River Type Segment Characteristics

Overall, the sinuous (Sn) River Type was the most abundant of the River Types in terms of total stream length, making 3915 km or 43 percent of entire Hunter network length (Table 4.3). The next most abundant was the high energy (HE) River Type, closely followed by the mid open valley mid energy (MOVME) River Type, comprising 2225 and 2165 km or 25 and 24 percent respectively. Seven percent of network length was made up of the constrained headwater (CHw) River Type, while the anabranching (Ab) River Type was the rarest in the Hunter, being only 65 km in length or less than one percent of the entire network.

The network of the Hunter River is comprised of 643 geomorphic river segments. HE River Types were the most abundant River Type in terms of total number of segments (200), marginally outnumbering Sn streams (197) (Table 4.3). MOVME and CHw River Types followed with a total of 123 and 118 segments respectively and Ab River Type was the least abundant with five individual segments in the entire network.

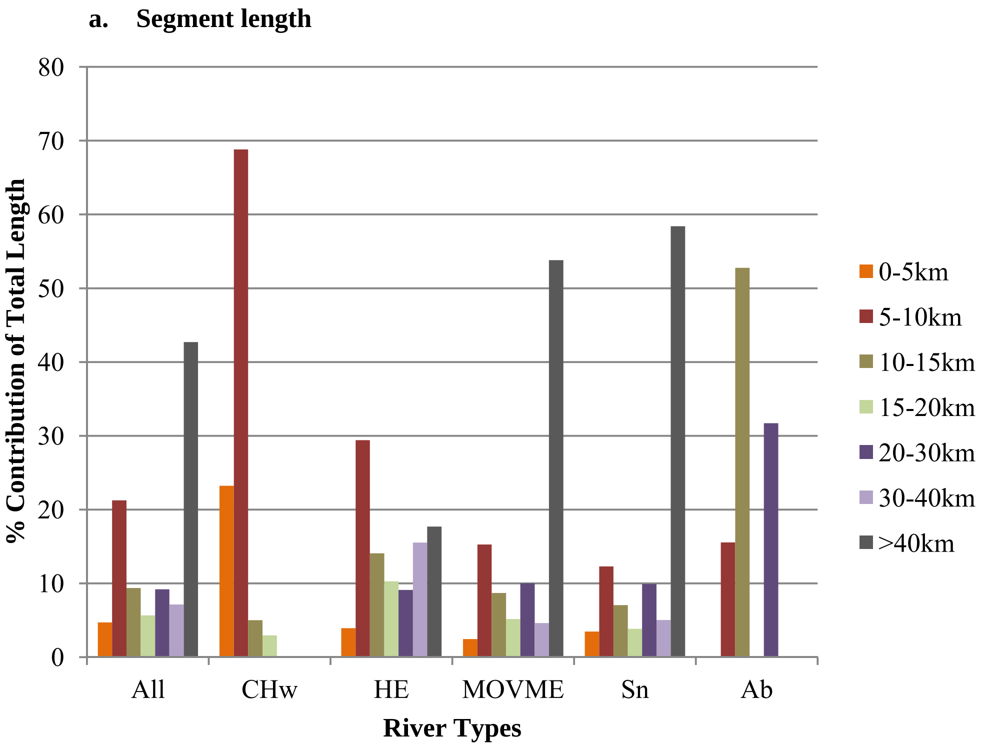
The average length of a continuous River Type segment was found to be 14.00 km per segment (Table 4.3). Sn River Type had the longest average segment length of 19.87 km, followed by MOVME River Type which was on average shorter by 2 km. The average segment lengths of Ab, HE and lastly CHw River Types, listed in descending order, were less than the entire network average. CHw River Types had the shortest average segment length of 5.38 km, less than half of the next shortest River Type at 11.13 km.

Table 4.3 Physical description of the five River Types in the Hunter catchment and their segment length and abundance characteristics.

River Type	Description	Abbreviation	Total Length (km)	% Total Length	Total Segments	Average Segment Length (km)
1	Constrained headwaters	CHw	635	7.1	118	5.38
2	High energy	HE	2225	24.7	200	11.13
3	Mid open valley moderate energy	MOVME	2165	24.0	123	17.60
4	Sinuous	Sn	3915	43.5	197	19.87
5	Anabranching	Ab	65	0.7	5	12.77
All			9005		643	14.00

Over 40 percent of the total Hunter network length was made up of continuous stream segments exceeding 40 km in length (Figure 4.7). Segments between the length of 5 km and 10 km were the next most abundant, comprising approximately 20 percent of the total network length. This was followed by segments of lengths between 10 and 15 km, 20 and 30 km, 30 and 40 km, 15 and 20 km and less than 5 km, listed in descending order, of which each segment length range made up less than ten percent of total network length. River Types MOVME and Sn exhibited similar segment length distributions to the entire Hunter network, over 50 percent of the total of each River Type length occurred as segments exceeding 40 km in length. Majority of total stream length of both CHw and HE River Types was made up of segments of length ranging between 5 km and 10 km. However, HE River Type was composed of a diverse range of segment lengths, unlike CHw where all segments were less than 20 km in length and 90 percent of stream length occurred as segments shorter than 10 km. Ab River Type occurred only in three segment length classes.

More than 50 percent of the segments in the Hunter River network are between 5 and 10 km in length (Figure 4.7b). This trend was evident in all River Types with segments of lengths in the range of 5 to 10 km being most abundant, other than with River Type Ab. The segment number frequency distribution of most of the River Types was unimodal, except River Types MOVME and Sn which had a bimodal frequency distribution. The major peak occurred at the 5 – 10 km interval and a second smaller peak occurred for segment intervals greater than 40 km in length.



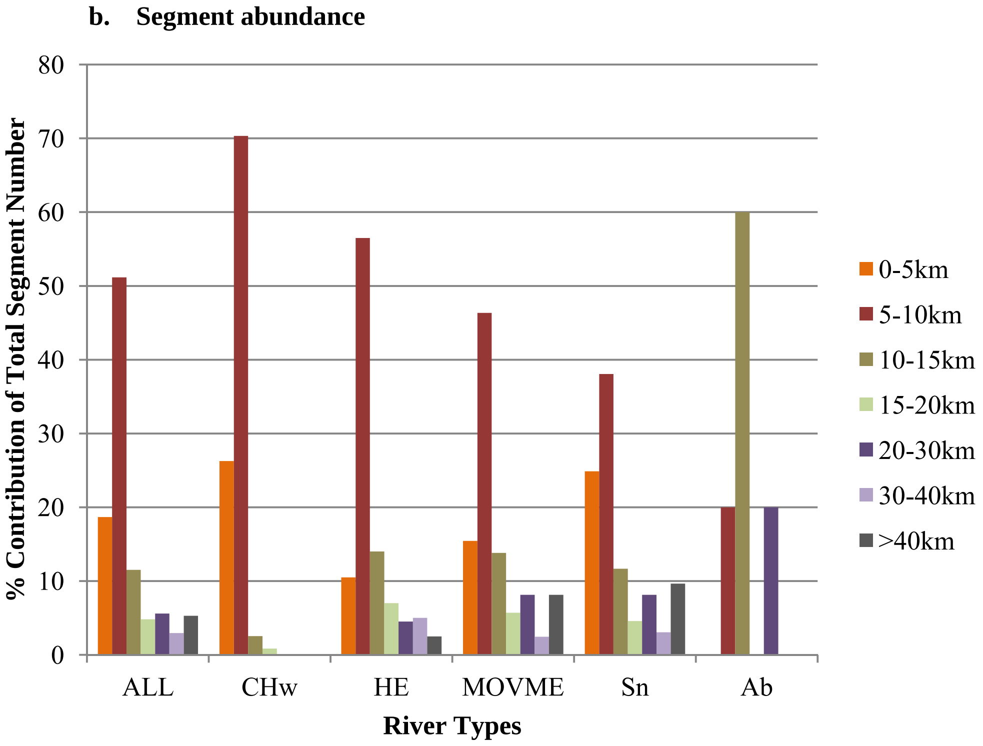


Figure 4.7 Distribution of a). Segment lengths; and, b). Segment abundance, for entire Hunter network and each individual River Type.

4.3.3 Anthropogenic Barriers

4.3.3.1 Barriers in the Hunter Basin

The Hunter River network contains 2466 anthropogenic barriers (Table 4.4). Road crossings were numerically dominant, representing 91.5 percent of all the barriers recorded in the network. This is followed by dams, with 193 identified structures, then weirs (10) making up less than one half a percent of all barriers. Floodgates were the least abundant of the barrier types, only six were identified on the Hunter network, making up less than a quarter of a percent of all barriers.

Broad spatial patterns in the distribution of barrier type were evident in the Hunter River network (Figure 4.8). Floodgates were only recorded on small tributaries in close proximity to the Hunter River estuary. Over 90 percent of all dams occurred in headwaters on first order streams, with only the larger reservoirs located further downstream. Weirs were generally situated on larger river channels and downstream of some of the major water reservoirs (dams), whereas road crossings were spread throughout the entire network of the Hunter River.

4.3.3.2 Spatial Distribution of Barriers

Distinct associations were present between barrier types and River Types in the Hunter River network (Table 4.4). All floodgates and weirs, with the exception of a single weir, occurred in MOVME and Sn River Types, of which the bulk of both barrier types (five of each) were located in MOVME River Type. HE and Sn River Types contained 156 of the 193 dams, of which majority of the barriers were situated in HE River Type. Road crossings were the only barrier type recorded within each River Type of the Hunter network and Ab River Type only contained road crossing type barriers. As a result, barrier distribution was not even between the five River Types of the Hunter River network, Distribution Indices (DI) varied between 0.32 and 1.10 (Table 4.4). River Types CHw and Ab, the least abundant of the River Types, also had the lowest barrier distribution within the network, each containing only approximately a third of the expected number of barriers (DI 0.32 and 0.34 respectively). Barriers were found to be the most evenly distributed within HE River Type, which had a DI value of 0.99. MOVME and Sn, the two most abundant lowland affiliated River Types within the Hunter catchment, also contained the

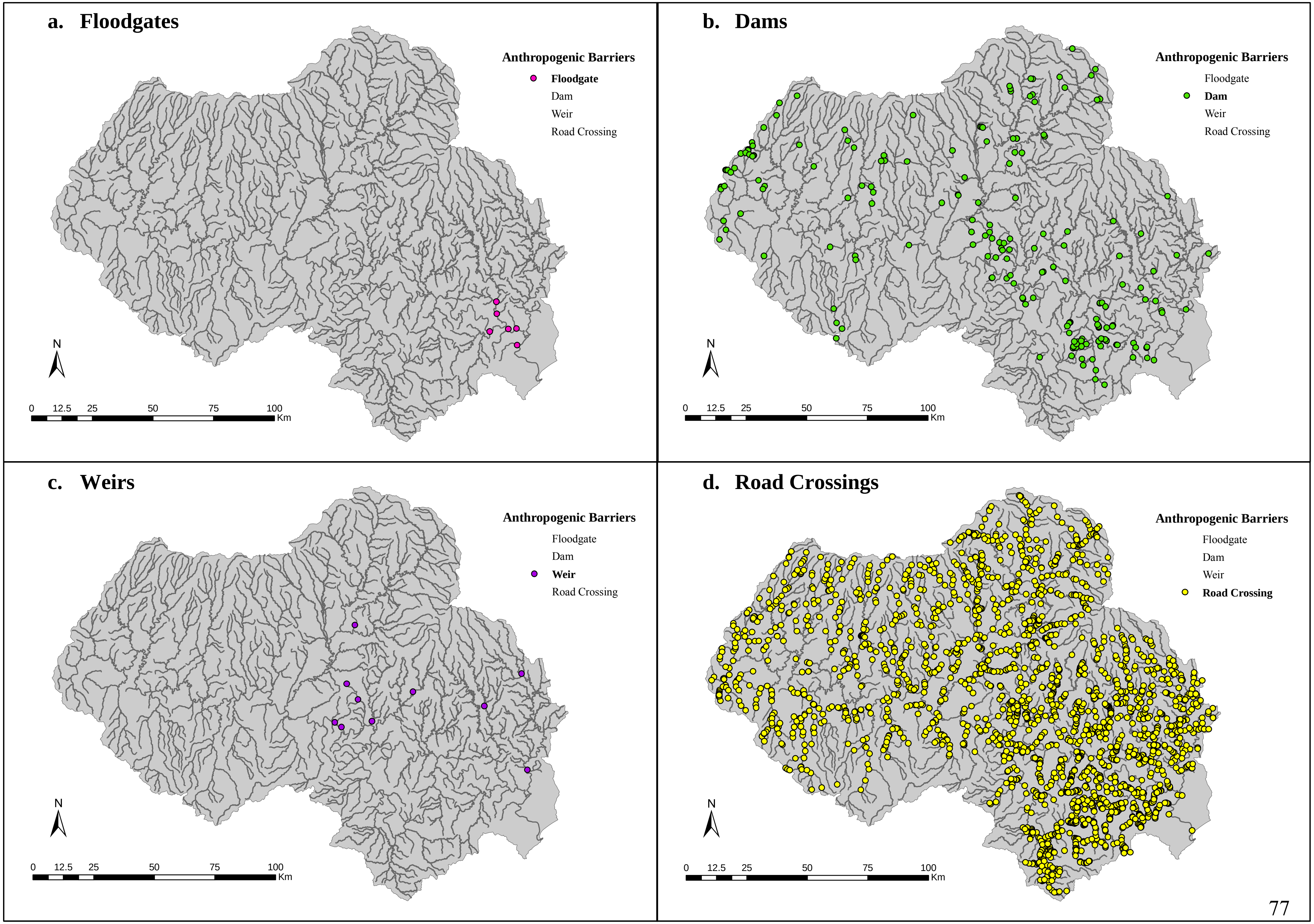


Figure 4.8 Spatial distribution of barriers within the Hunter River network a). Floodgates, b). Dams, c). Weirs; and, d). Road crossings.

largest number of barriers. The largest DI value of 1.10 occurred in MOVME streams, which was closely followed by Sn streams with a DI value of 1.07.

Table 4.4 Total number of each barrier type in the Hunter catchment and their distribution within each River Type. Proportion of total number of barriers within a River Type and proportion of total network comprised by each river type was standardised with the Distribution Index (D.I.) for each river type.

		Floodgate	Dam	Weir	Road Crossing	All Barriers	% Barriers	% Length	*D.I.*
River Type	**CHw**	0	10	0	45	55	2.23	7.05	***0.32***
	HE	0	85	1	515	601	24.37	24.71	***0.99***
	MOVME	5	27	5	615	652	26.44	24.07	***1.10***
	Sn	1	71	4	1073	1149	46.59	43.46	***1.07***
	Ab	0	0	0	6	6	0.24	0.71	***0.34***
	Other[1]				3	3	0.12		
All	**Total**	**6**	**193**	**10**	**2257**	**2466**			
	Percentage	*0.24*	*7.83*	*0.41*	*91.52*				

[1] Barriers falling on boundary of two River Types

4.3.4 Anthropogenic Fragmentation and Change in Stream Network Connectivity

4.3.4.1 Fragmentation of the Hunter River Network as a Continuum

4.3.4.1.1 Change in Network Connectivity and Segment Characteristics

The presence of anthropogenic barriers changed the connectivity and character of the Hunter River network. Before anthropogenic disturbance, only 14.5 km of total stream length was made up of segments less than 40 km in length (Figure 4.9). However, the addition of anthropogenic barriers within the Hunter River has resulted in 60 percent of the total network length of the Hunter River being dominated by stream segments of less than 10 km in length (Figure 4.9). Anthropogenic barriers also reduced the lengths of continuous stream segments from an initial network average of 1514 km per segment to only 3.7 km per segment (Table 4.5). Short segments less than 5 km in length made up around 80 percent of total segment number, whereas only 11 of the total 2475 segments were greater than 40 km in length (Figure 4.10; Table 4.5). The longest continuous segment of the naturally fragmented Hunter network comprised 98.3 percent of total stream length (Figure 4.11). Fragmentation because of the presence of barriers resulted in no continuous single stream segments constituting over 2 percent of entire network length, reducing length of longest continuous segment by 96 times.

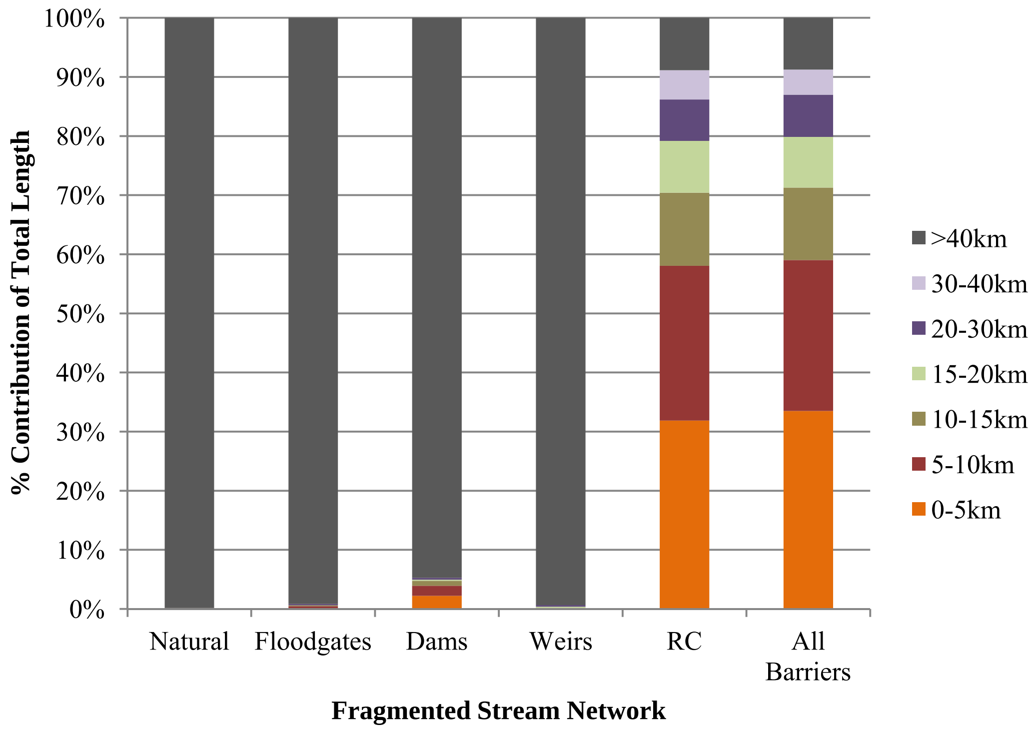


Figure 4.9 Distribution of segment lengths of Hunter River network when fragmented by floodgates, dams, weirs, road crossings and all barriers simultaneously.

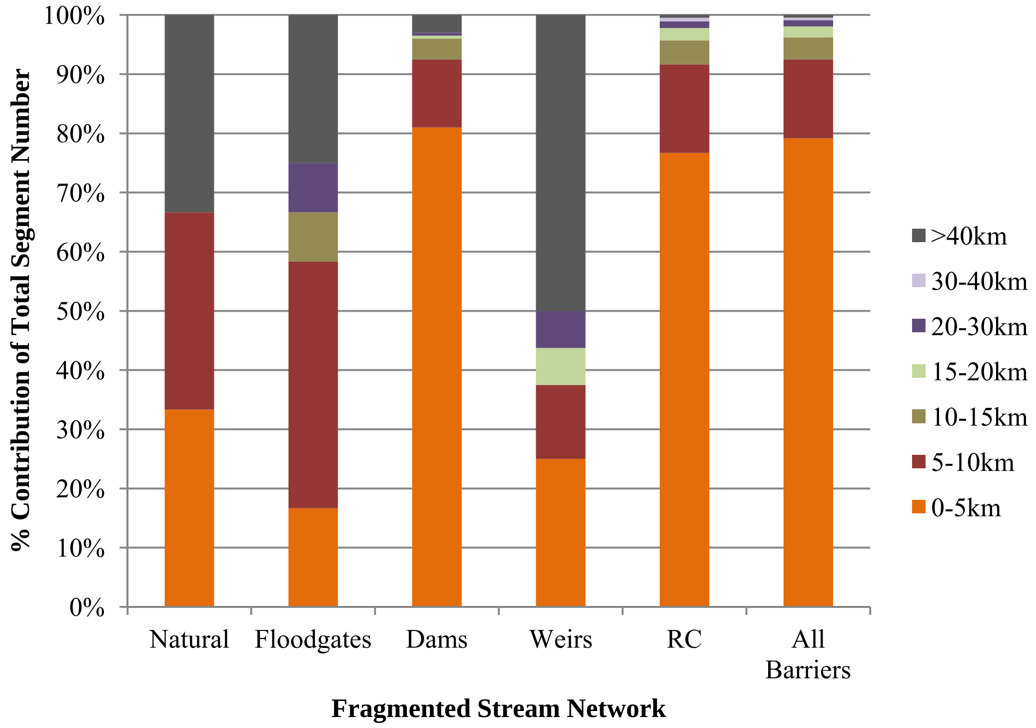


Figure 4.10 Distribution of segment abundance of Hunter River network when fragmented by floodgates, dams, weirs, road crossings and all barriers simultaneously.

The fully fragmented network is most similar in segment character to the stream network fragmented by road crossings, with difference in average length of continuous segments only varying by 0.34 km (Table 4.5). Both networks were dominantly characterised by an abundance of stream segments of short length and a scarceness of long continuous segments (Figure 4.9; Figure 4.10; Figure 4.11). On the other hand, over 95 percent of total Hunter River network length was comprised of continuous segments exceeding lengths of 40 km, when the network was only fragmented by floodgates, dams or weirs (Figure 4.9). Fragmentation due to floodgates and dams left 96 and 84 percent of Hunter stream length as a single continuous segment (Figure 4.11), with 75 and 97 percent of segments being less than 40 km long (Figure 4.10). However, the proportion of network comprised by the longest segment dropped to only 52 percent for the weir fragmented network, due to the central location of the ten weir-type barriers.

Table 4.5 Total number of segments and average length per segment of the Hunter River network as a result of each barrier type.

	Natural	Floodgates	Dams	Weirs	RC	All
Total Segments	6	12	199	16	2265	2475
Ave. Length (km)	1513.98	756.99	45.42	567.74	4.01	3.67

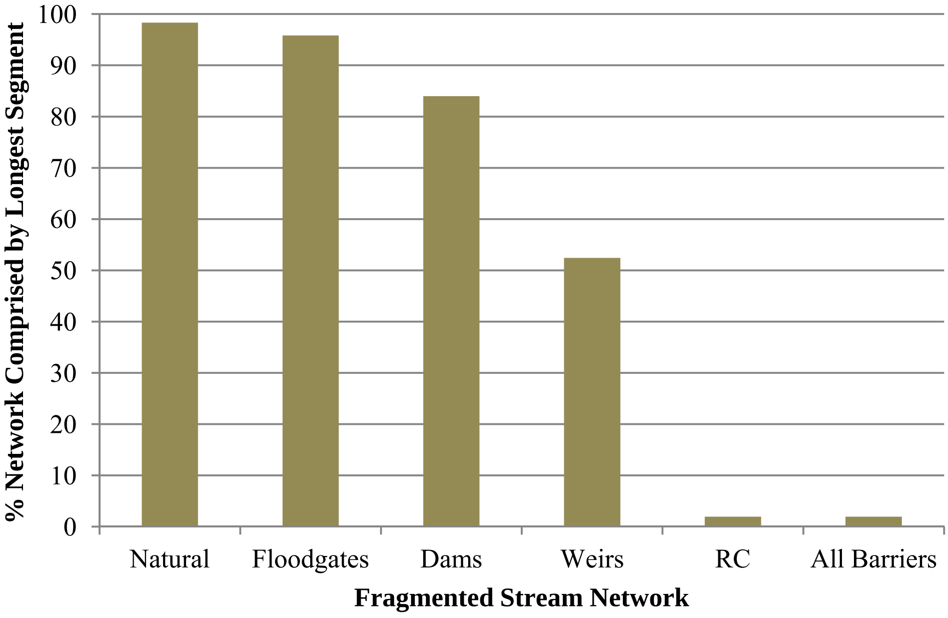


Figure 4.11 Percentage of the entire Hunter River network length which is comprised by the longest segment when fragmented by: floodgates, dams, weirs, road crossings and all barriers simultaneously.

4.3.4.1.2 Dendritic Connectivity Index

The Dendritic Connectivity Index for the Hunter River network decreased, for both diadromous (DCI$_D$) and potamodromous (DCI$_P$) migration patterns, as a result of network fragmentation by floodgates, dams, weirs, road crossings and all barriers combined. Barrier type effects on DCI values are listed in descending order in Table 4.6. Both DCI$_D$ and DCI$_P$ values of floodgate, dam or naturally fragmented networks were high and constantly in excess of 80 and 70 points respectively. Network connectivity levels decreased three-fold as a result of the presence of weirs, and decreased by 1277 (DCI$_D$) and 488 (DCI$_P$) times as a consequence of road crossings. Negligible differences in DCI values occurred when the network was fragmented by either road crossings or all barriers simultaneously, with change in DCI$_D$ and DCI$_P$ being zero and 0.004 respectively.

DCI$_D$ values were found to be larger than DCI$_P$ by up to 13 points, when the Hunter River network was both naturally fragmented and fragmented by floodgates or dams. However, when the Hunter River was fragmented by either weirs, road crossings or all barriers combined, DCI$_P$ exceeded DCI$_D$ between 0.12 and 7.6 points.

Table 4.6 Dendritic Connectivity Index (DCI) for diadromous and potamodromous species behaviour of each fragmented Hunter River network, listed in descending order.

		DCI	
		Diadromous	***Potamodromous***
Fragmented Network	***Natural***	98.332	96.715
	Floodgates	95.820	91.872
	Dams	83.986	70.980
	Weirs	29.122	36.739
	Road Crossings	0.077	0.198
	All Barriers	0.077	0.194

4.3.5 Fragmentation of the Hunter River Network as a Patch Mosaic

4.3.5.1 Segment Characteristics and Change in Connectivity

Fragmentation as a result of all barriers, changed the character of connectivity in the Hunter River network. At present the network is dominated by shorter segments, in terms of both total network length and segment length class abundance, with 73 percent of the network length being made up of segments shorter than 10 km (Figure 4.12f). The proportion of network length composed by segments up to 15 km in length, increased for the entire Hunter, and each individual River Type, in the range of 3 and 61.7 percent (Figure 4.12a and f), while largest percentage of network composed by a single segment dropped from 6.3 to 1.5 (Figure 4.15). The largest change in segment composition occurred with an increase in total number of segments less than 5 km in length, from 18.7 to 82.3 percent (Figure 4.13a and f). Connectivity of River Type CHw was least affected by fragmentation, with average segment lengths dropping from 5.38 km to 3.67 km (Figure 4.14 - CHw). However, River Type Ab was composed of the longest continuous segments averaging at 5.8 km. River Types HE, MOVME and Sn had a mean length of around 2.9 km, equal to the entire Hunter River network segment average. No segments longer than 30km were found in River Type HE, as a result of fragmentation by all barriers (Figure 4.12f).

Change in connectivity of the Hunter River network as a result of all barriers was most similar to that of the network fragmented by road crossings only. Size of the longest continuous segment of the entire network and each individual River Type, other than CHw, were wholly determined by road crossings (Figure 4.15). However, it was found that the size of the longest continuous segment of the entire Hunter River network and River Types MOVME and Sn, were also substantially reduced by weirs, from 6.3 to 4 percent, 26.1 to 16.6 percent and 11.3 to 8 percent respectively. Difference between the mean continuous segment length of the Hunter River network fragmented by all barriers and road crossings varied up to a maximum of 0.33 km per HE River Type segment (Figure 4.14). Fragmentation as a result of dams increased the proportion of segments less than 5 km in length to 39.5 percent (Figure 4.13c), while simultaneously reducing mean segment length to 10.8 km (Figure 4.14).

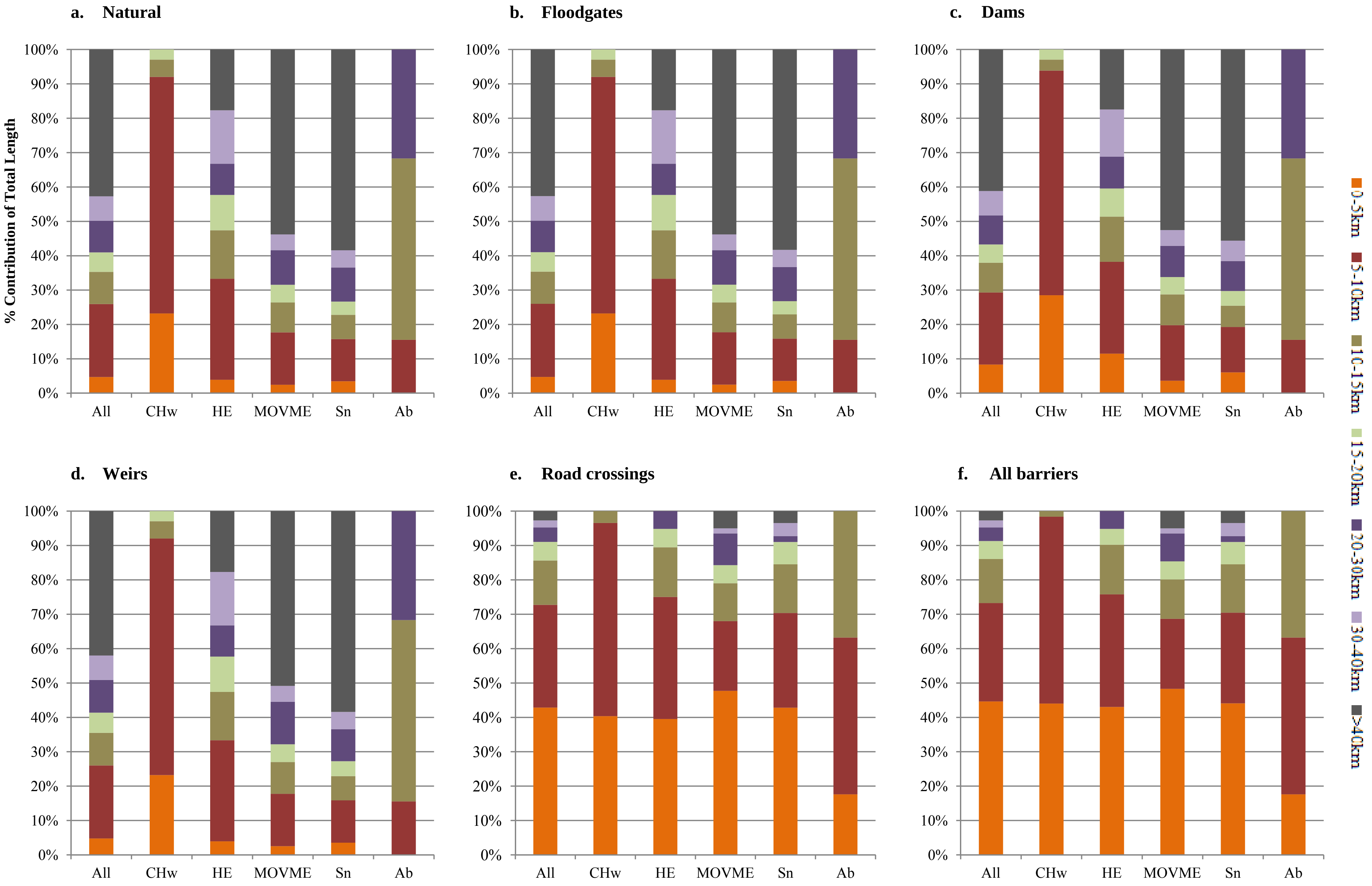


Figure 4.12 Distribution of stream segment lengths of Hunter River network and each individual River Type fragmented by a). Natural, b). Floodgates, c). Dams, d). Weirs, e). Road crossings; and, f). All barriers.

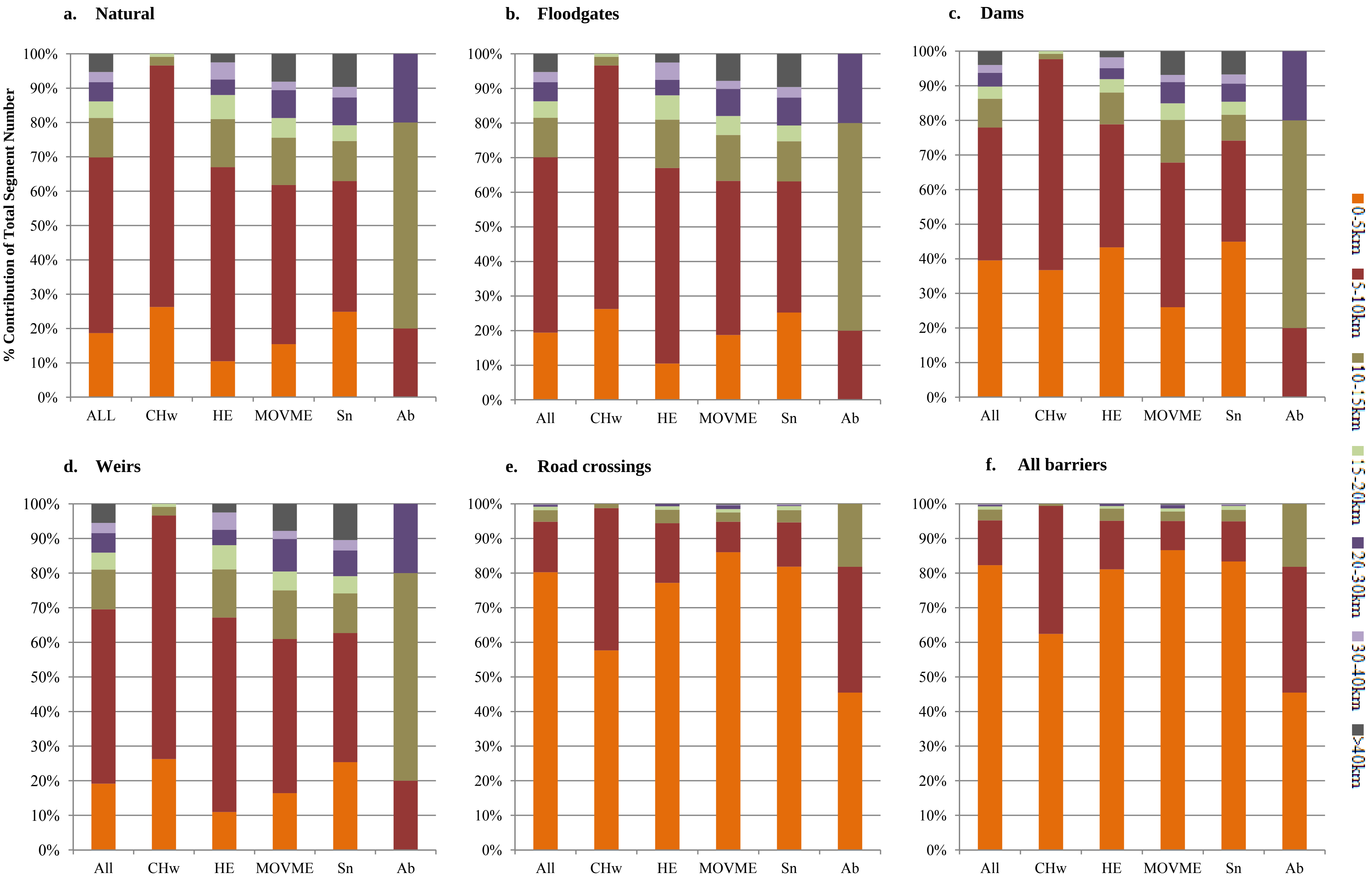


Figure 4.13 Distribution of number of stream segments of Hunter River network and each individual River Type fragmented by a). Natural, b). Floodgates, c). Dams, d). Weirs, e). Road crossings; and, f). All barriers.

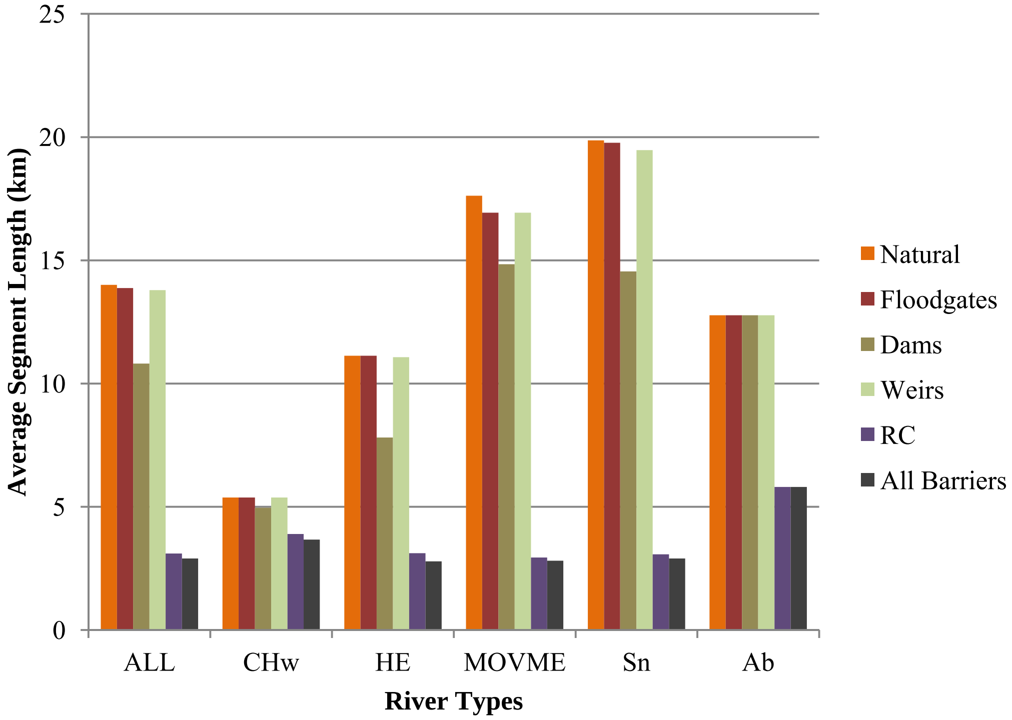


Figure 4.14 Average segment length of Hunter River network and each individual River Type, when network is fragmented by floodgates, dams, weirs, road crossings and all barrier types simultaneously.

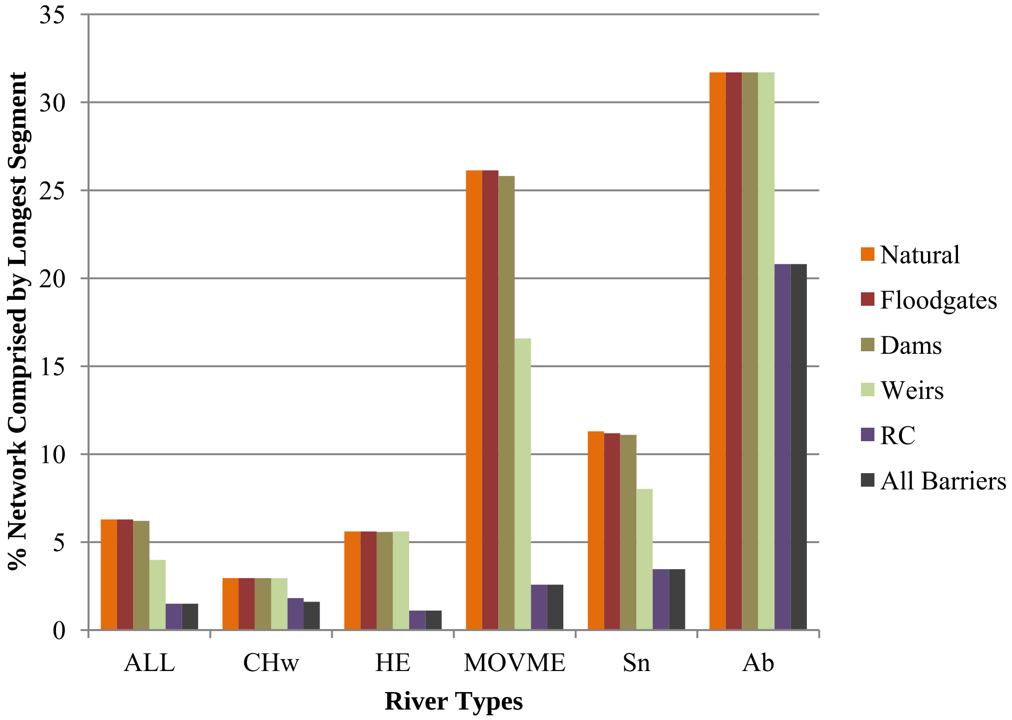


Figure 4.15 Percentage of the Hunter River network and each individual River Type comprised by the longest respective segment, when fragmented by floodgates, dams, weirs, road crossings and all barriers combined.

4.3.5.2 Change in Stream Network Character

The physical character of the Hunter River network, fragmented by road crossings only and that by all barrier types, is statistically different to the natural, floodgate, and weir and dam fragmented Hunter River networks (Table 4.7). However, the same level of dissimilarity (0.502) was present between the naturally fragmented network and networks fragmented by road crossings and all barriers simultaneously. Largest dissimilarity (0.512) of Hunter River network character occurred between the floodgate and road crossing fragmented networks. Floodgate, weir and dam, and naturally fragmented Hunter River networks had statistically similar stream segment character.

In addition, the Hunter River network fragmented by road crossings had the lowest variability, or dispersion, between stream segment characteristics of the five River Types, with an MDI value of 0.58 (Table 4.8), which was closely followed by the network fragmented by all barriers combined (0.604) and then weirs and dams (1.157). Dispersion of River Type characteristics were highest within the naturally fragmented network, with an MDI value of 1.343. Furthermore, fragmentation as a result of floodgates resulted in the smallest reduction in River Type dispersion of any artificially fragmented Hunter River network (0.027 point drop in MDI to 1.316), indicating that floodgates minimally altered the character of the network and higher levels of variability of network character were maintained.

Table 4.7 ANOSIM results (Gower Metric) comparing similarity of all networks to each other. Statistically different networks are highlighted in yellow (value > 0.3).

	Natural	**All Barriers**	**Road Crossings**	**Floodgates**	**Weirs & Dams**
Natural					
All Barriers	0.508				
Road Crossings	0.508	-0.186			
Floodgates	-0.2	0.508	0.512		
Weirs & Dams	-9.6E-02	0.358	0.332	-0.108	

Table 4.8 Multivariate Dispersion Indices for each network type, listed in ascending order.

Network	Dispersion
Road Crossings	0.58
All Barriers	0.604
Weirs and Dams	1.157
Floodgate	1.316
Natural	1.343

4.4 DISCUSSION

4.4.1 Anthropogenic Barriers of the Hunter River and their Spatial Distribution

The longitudinal connectivity of the Hunter River network has been fragmented because of the presence of 2466 anthropogenic barriers. Previous studies in the Hunter catchment have evaluated barrier abundance, with particular interest into which and how many structures acted as barriers to fish migrations. The numbers of identified barriers have varied between 18 (Harris, 1984) to 323 barriers (Thorncraft and Harris, 2000), both of which are significantly less than the number of barriers identified in this thesis. There are two main probable reasons for the differences in results. Firstly, it could be attributed to the selective inclusion of only certain barrier types in previous studies, with Harris (1984) only considering large impoundments (large dams, weirs and tidal barriers) while Thorncraft and Harris (2000) did not include numerous small structures (e.g. road crossings) in their analysis. Other studies, such as those of Williams and Watford (1997) and Gordos *et al.* (2007), identified abundances of smaller barrier types in the region, however these studies were carried out over larger geographical scales (all NSW east coast basins) which are incompatible with the spatial limits of this study. This thesis includes all in-stream structures which potentially act as barriers to the movement of water, sediment, nutrients and aquatic organisms, and thus may represent an overestimation of the number of barriers in the Hunter River.

The abundance of structures varied significantly between the four identified barrier types in the Hunter River network. Road crossings were the predominant barrier type, making up over 90 percent of all structures in the region (Table 4.4). This appears to be a common attribute across all coastal drainage basins in NSW (Gordos *et al.*, 2007), and is most likely a combined consequence of both high population density in the area, and due to the necessity and ease of construction of these small commonly unregulated structures (unlike with larger, complex and costlier barriers exemplified by dams). By comparison, low head barriers such as weirs and milldams have been reported to be abundant barrier types in various northern hemisphere river systems, where many of the barriers had been constructed more than a century ago so as to divert water to mills (Garcia de Leaniz, 2008; Walter and Merritts, 2008). Understanding these variations in barrier abundance is significant and an aspect that needs to be considered in the conservation of Australian streams, especially as the evidently high numbers of smaller barriers can have substantial cumulative impacts on the river system, which are potentially comparable to larger anthropogenic structures that are significantly better documented (Harris, 1984; Arthington and Pusey, 2003).

The distribution of the four individual barrier types differed between the five identified geomorphic River Types of the Hunter River network. Most of the barrier types had a predominant association with a specific River Type (Table 4.4), other than road crossings which were spread across the entire network and all River Types. The distribution of various other dams and weirs has been closely associated with steep tributaries (Garcia de Leaniz, 2008) and low order streams (Anderson *et al.*, 2008; Walter and Merritts, 2008) where channel slope was a significant factor in the choice of barrier sites. These observations are comparable to the results obtained in this thesis, where approximately half of all dams were located in the steeper headwater segments of the stream network (CHw and HE River Types). In addition, Park *et al.* (2008) found that culverts only occurred on streams that were an order four or lower.

The associations between barrier type and River Type may result in unique interactions and barrier-specific degradation of stream morphology and the respective ecosystems. For example, each individual River Type can have unique physical characteristics and riparian ecosystems (Scown, 2010) that could be distinctly compromised by the associated anthropogenic barriers, such as dams, which are known to affect both pattern and process of river systems (Jansson *et al.*, 2000; Walter and Merritts, 2008). Quantifying these potentially exclusive interactions is significant to river conservation so as to allow river scientists and managers to predict and consequently prevent destruction of stream habitat heterogeneity. In addition, barrier types can have more adverse effects on some sections of river systems than on others (Park *et al.*, 2008; Poplar-Jeffers *et al.*, 2009). Identifying the distribution of structures of a barrier type within a stream network can facilitate understanding of which regions are under greater anthropogenic stress, while also providing a tool to reduce impacts of future developments by changes to barrier design or through the avoidance of certain sensitive River Types.

Anthropogenic barriers are unevenly distributed throughout the five geomorphic River Types of the Hunter River network. The abundant lowland-associated River Types (Table 4.4 - MOVME and SN) had the largest proportion of barriers and, as a result, were the most heavily impacted geomorphic units in the stream network. In contrast, River Types CHw and Ab had the lowest barrier distribution and were the least impacted. No studies have previously evaluated the spatial distribution of anthropogenic barriers within a stream network. Dynesius and Nilsson (1994) and Nilsson et *al.* (2005) investigated the degree to which large river catchments worldwide have been impacted as a result of fragmentation and flow regulation by dams. Catchments were classified as unaffected, moderately affected or strongly affected. Fragmentation categories were based upon whether the dams were located on a minor tributary, major tributary or main channel of the system, and how far downstream on the main channel the dam occurred. Their studies highlighted the variability between levels of disturbance of entire catchments at continental scales. Australasia was the least effected continental region, with 74 percent of river systems categorised as unaffected (Nilsson *et al.*, 2005). On the other hand, European river systems were the most heavily impacted with 63 percent categorised as highly affected. River systems of Africa and Asia were evenly distributed between the three impact categories. These studies and the findings of this thesis highlight the uneven distribution of anthropogenic barriers, and thus the extent of disturbance, across riverine landscapes at different spatial scales.

The uneven distribution of barriers throughout the stream network could have implications on the quality and availability of habitats, and their respective ecosystems. Regions of the stream network that have a larger proportion of anthropogenic barriers are potentially undergoing change and degradation at a greater rate than regions with lower barrier abundance. In the coastal catchments of NSW, downstream reaches support fish communities with higher species richness (Gehrke and Harris, 2000). These communities could be under greater threat than the more distinct montane fish communities due to the increased levels of disturbance to the lowland River Type associated habitats. Quantifying the distribution of barriers through a stream network elucidates regions which are at greater risk of alteration and loss, and can guide management actions to restore and protect threatened streams and habitats.

4.4.2 Anthropogenic Change in Stream Network Connectivity of the Hunter River

Stream network connectivity, based on the conceptualisation of the river system as a continuum, decreased substantially due to fragmentation by anthropogenic barriers. Naturally, the stream network was composed of a few long continuous segments. However, as a result of anthropogenic fragmentation the character of the system shifted towards dominance by short disjunct segments, in terms of both the segment length and segment number (Figure 4.9; Figure 4.10; Figure 4.11), with network connectivity decreasing substantially (Table 4.6). In the mid-Atlantic region of the USA, Walter and Merritts (2008) found that streams were highly fragmented by milldams, with the typical spacing between barriers being in the range of 2.4 and 5 km along the stream. This is a similar order of magnitude to the results obtained in this thesis (Table 4.5).

Australian (Harris, 1984) and international (Dynesius and Nilsson, 1994; Nilsson *et al.*, 2005; Anderson *et al.*, 2008; Park *et al.*, 2008) studies have highlighted that stream network fragmentation is widespread. The extent of fragmentation between catchments has been variable, with certain catchments being highly impacted and others less so (Dynesius and Nilsson, 1994; Nilsson *et al.*, 2005). The findings of this thesis suggest that the Hunter River network is more fragmented by anthropogenic barriers than numerous other Australian and international river systems (Table 4.9), with higher amounts of the stream network located upstream of anthropogenic barriers. Only five of the other 21 coastal catchments in NSW and south-eastern Queensland examined by Harris (1984) had a larger proportion of network located upstream of barriers than found in this thesis for the Hunter River (excluding road crossings). These catchments were the Mary River, Brisbane River, Macquarie and Tuggerah Lakes, Sydney Coast and Shoalhaven River. Furthermore, Nilsson *et al.* (2005) reported that of the 292 large river systems in the world, 52 percent were fragmented by dams on the main river channel, 59 percent on a tributary and 35 percent of large river systems were unfragmented by dams. Dams fragment both the main river channel and multiple tributaries of the Hunter River network, thus categorising it as either a moderately or strongly effected river system (Dynesius and Nilsson, 1994).

Table 4.9 A comparison of the extent of anthropogenic fragmentation of the Hunter River network and other stream networks.

River System	Total Percent of Network above Barriers	Types of Barriers	Reference
This thesis			
Hunter River, Australia	4.18	Floodgates	
	16.01	Dams	
	70.88	Weirs	
	99.92	Road crossings	
	99.92	All	
Other Studies			
Hunter River, Australia	29	Weirs, dams	Harris (1984)
NSW Coastal Catchments, Australia	32	Weirs, dams, tidal barriers	Harris (1984)
Swan River, Alberta, Canada	19.9	Hanging culverts	Park *et al.* (2008)
Notikewin River, Alberta, Canada	5.4	Hanging culverts	Park *et al.* (2008)
Calling River, Alberta, Canada	4.8	Hanging culverts	Park *et al.* (2008)
Christina River, Alberta, Canada	4.5	Hanging culverts	Park *et al.* (2008)
Sarapiqui River, Costa Rica	9.4	Dams	Anderson *et al.* (2008)
Upper Cheat River, West Virginia USA	33	Culverts	Poplar-Jeffers *et al.* (2009)[1]
Willamette and Lower Columbia River, Washington and Oregon, USA	42	Culverts, dams, natural and further unknown	Sheer and Steel (2006)[2]

[1] Percent of network refers to Brook trout habitat located upstream of impassable culverts

[2] Study includes all known in-stream barriers in analysis, without explicitly defining all of the barrier types

A previous study on the fragmentation of the Hunter River network was carried out by Harris (1984). The study identified that approximately 29 percent of the stream length was restricted from upstream migrations by artificial barriers (Table 4.9). This thesis found that restrictions to upstream migrations were significantly higher (Table 4.6; Table 4.9). These discrepancies are potentially a combined consequence of the limited number of barriers incorporated into the analysis by Harris (1984) and the lack of differentiation between potential and actual barriers to upstream movement in this thesis.

Fragmentation by anthropogenic barriers also decreased the connectivity of the river system when based on a mosaic of River Types. Similar trends were evident between both representations of the Hunter River network, with the character of the system shifting from predominantly long continuous segments to plentiful shorter fragmented segments (Figure

4.12; Figure 4.13; Figure 4.15). Even though the importance of spatial arrangement of stream networks on riverine patterns and processes (Power and Dietrich, 2002; Benda *et al.*, 2004; Campbell Grant *et al.*, 2007) and the impact anthropogenic barriers such as dams have on stream morphology and habitat availability is well appreciated (Petts, 1984; Erskine, 1985), the change in character and spatial distribution of geomorphic units at a basin scale is still relatively unexplored in the literature. However, a number of studies have examined catchment-scale fragmentation and loss of riverine habitats by artificial barriers (Sheer and Steel, 2006; Anderson *et al.*, 2008; Poplar-Jeffers *et al.*, 2009). Anderson *et al.* (2008) found that certain habitat types in Costa Rica were more fragmented than others, resulting in the uneven loss of different habitats in a river system. This is comparable to the findings in this thesis, where certain River Types (such as MOVME and Sn) were more fragmented by artificial barriers than others (such as CHw and Ab).

The extent of influence each type of barrier had on the current state of stream network connectivity of the Hunter River network varied between barrier types. The road crossing barrier type had a substantial influence on the current state of stream network connectivity, largely determining all segment and connectivity characteristics for both physical representations of the stream network (Table 4.6; Table 4.7; Table 4.8). On the other hand, the natural state of connectivity was little altered by floodgates. Dams and weirs influenced different aspects of connectivity of the Hunter River network (Figure 4.14; Figure 4.15; Table 4.6 and others), although the extent of deviation from the natural state was far less substantial than that for road crossings (Table 4.7; Table 4.8). These findings suggest that road crossings have a substantial influence on anthropogenic change to stream network connectivity, such as that highlighted by Park *et al.* (2008) and Poplar-Jeffers *et al.* (2009). To date, most stream fragmentation studies have been focused on a single barrier type (e.g. Anderson *et al.*, 2008; Park *et al.*, 2008; Walter and Merritts, 2008; etc.; Poplar-Jeffers *et al.*, 2009). However, the results of this thesis suggest that cumulative impacts of multiple barrier types, rather than individual types, determined the final state of connectivity of the Hunter River network. All agents of change should be considered when assessing stream fragmentation so as to ensure a complete understanding of the system without bias towards a single type of obstruction.

Stream network connectivity and character of the Hunter River was influenced by two primary factors. Fragmentation of the stream network generally increased with an increase in barrier abundance. For example, connectivity of the Hunter River network had decreased most substantially due to road crossings, which were the most abundant barrier type in the region. These findings are comparable to other studies such as that of Park *et al.* (2008), where the most fragmented sub-catchment had the highest number of impassable culverts. However, the extent of fragmentation was not only influenced by barrier abundance but also barrier location in the stream network. Most notably, stream network connectivity was substantially lower when the Hunter River was fragmented by weirs rather than dams (Figure 4.11; Figure 4.15; Table 4.6), even though the number of weirs was approximately a twentieth of the dam total. This was a result of weirs being located on the main channels and in more central regions of the Hunter River network, unlike dams which were predominantly in headwater streams (Figure 4.8). These outcomes are consistent with other research which has emphasised the importance of barrier location on river system structure and function, where stream connectivity decreased due to central and downstream barrier locations (Dynesius and Nilsson, 1994; Nilsson *et al.*, 2005; Cote *et al.*, 2009; Rolls, 2011). Barrier location is an aspect that should also be incorporated in future development planning, so as to minimise negative impacts on stream systems and protect large tracts of undisturbed and connected river which are ecologically important (Morita and Yamamoto, 2002; Perkin and Gido, 2011).

Anthropogenic changes in stream network connectivity of the Hunter River have potentially degraded the structure and function of the river system and its respective ecosystems. The Hunter River network provides an assortment of habitats which support a number of distinct riverine fish communities (Gehrke and Harris, 2000) and invertebrates (Ruello, 1973). The diverse fish species have different behaviour, with 24 undergoing a type of migratory movement as part of their life cycle (Appendix A). The results of this thesis suggest that anthropogenic barriers may have reduced stream network connectivity by up to 1277 times for diadromous behaviour and 499 times for potamodromous behaviour (Table 4.6). These changes to stream network connectivity may have restricted mobility of aquatic biota between critical habitats. Australian bass, long-finned eel, striped gudgeon and Australian smelt are examples of fish species with different migratory requirements that may be severely affected by decreases in connectivity.

In the Hunter River and temperate coastal catchments in south-eastern Australia, the repercussions of anthropogenic barriers are already evident. In particular, a decrease in the number and abundance of migratory species and shift towards habitat generalists has been noted upstream of artificial barriers (Gehrke *et al.*, 2002; Brainwood *et al.*, 2008; Rolls, 2011; Rolls *et al.*, 2011). Other negative implications of fragmentation and decreased habitat size have been identified worldwide, such as local extinctions, reduced genetic diversity (population fitness) and species extinctions. Similar or even more severe processes due to changes in stream network connectivity may be influencing Australian freshwater communities. Endemic species generally have weaker swimming and jumping abilities in comparison to their northern hemisphere counterparts, which may be further increasing their susceptibility to anthropogenic fragmentation. Further research on the associations between structural changes in stream network connectivity and ecological structure and function could provide valuable information to managing and conserving the Hunter catchment and its ecosystems.

4.5 CHAPTER SUMMARY

Network connectivity within the Hunter River substantially decreased as a result of anthropogenic fragmentation. The character of the river system shifted from dominance by a few long continuous stream segments to a system composed of a high number of short discontinuous stream segments. These trends were pronounced for both representations of the Hunter River network, namely the river system as a continuum and a mosaic of River Types, and could have major implications on riverine processes and integrity of the local ecosystems.

Anthropogenic barriers were unevenly distributed throughout the stream network. As a result, certain components of the stream network were more significantly impacted by anthropogenic fragmentation than others, with a larger conglomeration of barriers in the lowland streams than the headwater-associated river types. In addition, the uneven distribution of barriers was critical in determining the extent of change in connectivity of the Hunter River network. Barriers located more centrally and lower in the network disrupted stream network connectivity of the entire river system more widely than barriers which were situated further upstream in headwater regions of the network. A number of different barrier types were responsible for fragmenting the Hunter River network. The types of barriers included weirs, dams, floodgates and road crossings. Each of the barrier types except for the latter was predominantly associated with a certain section of the stream network, potentially resulting in unique (barrier-associated) interactions and stream degradation. Furthermore, road crossings were the probable main agents of fragmentation in the Hunter River network, due to their high abundance and widespread distribution, which was incomparable with any of the other types of barriers. This thesis highlights the potential significance of smaller structures such as road crossings in fragmenting stream networks, rather than just focusing on the larger and more pronounced barrier types.

In the following chapter, an optimisation model for restoring stream network connectivity through barrier modifications is developed and the sensitivity of the model is explored using the Williams River catchment (a major tributary of the Hunter River). As shown in this chapter, the Hunter River is a system highly fragmented by anthropogenic barriers. Consequently, the stream segment and barrier data collected in this chapter are applied in the model building and sensitivity analysis carried out in the following chapter.

Chapter 5

5 Chapter 5 – Optimising Restoration of Stream Network Connectivity

5.1 INTRODUCTION

Human induced change of the natural environment has degraded riverine landscapes (Nilsson *et al.*, 2005). Disruptions of natural connections within river ecosystems by the presence of artificial barriers is widespread (Harris, 1984; Dynesius and Nilsson, 1994; Nilsson *et al.*, 2005) and represents a substantial threat to freshwater ecosystem integrity and resilience (Ward and Stanford, 1995; Gehrke *et al.*, 2002; Pringle, 2003). Riverine landscapes, their ecosystems and the goods and services they provide are highly valued (Costanza *et al.*, 1997; Thoms and Sheldon, 2000) and substantial effort is being placed into conserving intact natural environments as well as restoring degraded systems. There has been a paradigm shift in river restoration over the last decade; moving from engineering-dominated solutions to ecosystem approaches (Hillman and Brierley, 2005). This has been accompanied by an emphasis on restoring riverine landscapes within a catchment context and the preferential restoration of process over pattern (Sear, 1994; Bohn and Kershner, 2002; Clarke *et al.*, 2003; Lake, 2005; Wohl *et al.*, 2005; Kondolf *et al.*, 2006; Roni *et al.*, 2008; Beechie *et al.*, 2010). Riverine restoration has become a significant industry, with annual expenditure often exceeding a billion dollars in some countries (Bernhardt *et al.*, 2005; Alexander and Allan, 2006; Brooks and Lake, 2007). However, projects are frequently implemented at local spatial scales with limited coordination between multiple restorations or awareness of their role in a catchment context. The need for systematic restoration planning to improve the allocation of funds by incorporating cumulative ecological and economic tradeoffs has been identified as an area of concern (Lake, 2005; Thorp *et al.*, 2008; Hermoso *et al.*, 2012).

This chapter details the development of an optimal restoration approach for improving stream network connectivity. It does so by identifying the anthropogenic in-stream barriers which, if physically modified, achieve maximum connectivity improvement for an entire stream network. The approach taken includes a specifically built optimisation model that identifies the best combination of barriers within user-specified budgetary restrictions, where the level of stream network connectivity is evaluated via the dendritic connectivity index (Cote *et al.*, 2009). The chapter explores to what extent the character of the stream network and the presence of in-stream barriers influence restoration of connectivity. The importance of species behaviour (migration type) on restoring connectivity is also assessed along with the impact of sequentially selecting barriers on the level of improvement in connectivity. A series of sensitivity analyses of the model are carried out on the Williams River sub-catchment, a major tributary of the Hunter River. The research undertaken in this component of the thesis provides the basis for Chapter 6, by establishing the importance of numerous stream network and barrier characteristics in the barrier selection and connectivity restoration procedure, which is applied to the entire Hunter River catchment.

5.2 METHODS

5.2.1 Introduction

The methods employed to build and apply the restoration optimisation model to maximise stream network connectivity is described in four sections. The first defines the stream network connectivity restoration problem as a mathematical model and explains how a genetic algorithm can be used to solve it. Section two describes the data sources and methods used to extract data that are used as inputs to the mathematical model. The third section outlines a sensitivity analysis of the model and the sequential optimisation procedure used. Finally in section four, the techniques employed in the analysis of data are discussed. All modelling and optimisation work was carried out with *MATLAB R2011b* software package.

5.2.2 Problem Definition and Optimisation Model

Stream network connectivity does improve with the removal of anthropogenic barriers (Catalano *et al.*, 2007) and, or alterations to the barrier itself (Boys *et al.*, 2012; David and Hamer, 2012). This section mathematically defines the model which is developed to assist in maximising stream network connectivity, through identifying the optimal combination of barriers to be physically modified for the maximum possible network connectivity within a specific budgetary limit. In this thesis, barrier modification refers to any activity that improves porosity of a barrier or increases the probability for species to be able to move past it. Barrier removal, installation of fishways, installation of baffles and barrier operation management are all examples of barrier modification, in addition to numerous other activities. Thus, the modification of barriers results in the improvement or restoration of stream network connectivity, and the two activities are synonymous with each other.

The Dendritic Connectivity Index (DCI) is a single quantitative measure of stream network connectivity that has been successfully used by Cote *et al.* (2009) as an objective function and is used in this thesis to maximise the problem formulation. The original DCI exists in two forms, for a diadromous and potamodromous network connectivity application (Equation 5.1; Equation 5.2). Consequently, two basic stream network connectivity objectives can be derived from the DCI. In this study the two network connectivity objectives were unified into a single weighted multi-objective equation (Equation 5.3), which allowed for either or both of the migratory types to be targeted by placing the desired weighting ($0 \leq v \leq 1$; Equation 5.3) onto the two forms of species migration. For example, connectivity would be restored for diadromous migration only with a weighting of one, for potamodromous migration only with a weighting of zero, for both migration types equally with a weighting of 0.5, and so on. The use of complementary weightings on multiple objectives is a technique commonly applied in optimisation problems to incorporate and merge multiple objectives into a single objective function (e.g. Kuby *et al.*, 2005; Zheng *et al.*, 2009). This allows the user to weight each objective as desired.

The DCI used in this thesis, was altered in quantifying connectivity so as to incorporate habitat quantity and quality because both have been identified as critical attributes of riverine landscapes (Wiens, 2002). The stream segment length variable was altered to allow for weighting of segment lengths based on the quality of segment stream type (Equation 5.4; Equation 5.5; Figure 5.1). This is a commonly utilised technique in spatial graphing (Proulx *et al.*, 2005). DCI is a function of stream segment size and the probability of movement, in both upstream and downstream direction between either the ocean and stream network segments (diadromous species behaviour) or movement between different segments within a stream network (potamodromous species behaviour). Stream segment lengths are constants in the objective function of a specific problem. However, the

$$DCI_D = \sum_{i=1}^{n} c_{ij} \frac{l_i}{L_{TW}} * 100$$

Equation 5.1 Modified Dendritic Connectivity Index incorporating stream network habitat quality weighting: Diadromous application

$$DCI_P = \sum_{i=1}^{n} \sum_{j=1}^{n} c_{ij} \frac{l_i}{L_{TW}} \frac{l_j}{L_{TW}} * 100$$

Equation 5.2 Modified Dendritic Connectivity Index incorporating stream network habitat quality weighting: Potamodromous application

$$Z = v.DCI_D + (1 - v).DCI_P$$

Equation 5.3 Multi-objective function, incorporating both modified diadromous and potamodromous network connectivity objectives, where v is weighting on DCI_D, and $0 \leq v \leq 1$

$$l_i = \sum_{z=1}^{T} w_z \, l_{iz}$$

Equation 5.4 Quality weighted stream segment length

$$L_{TW} = \sum_{z=1}^{T} w_z \, L_z$$

Equation 5.5 Quality weighted stream length of entire network

$$c_{ij} = \prod_{m=1}^{M} (p_m^u \, p_m^d + \Delta p_m . X_m)$$

Equation 5.6 Cumulative barrier porosity between i^{th} and j^{th} segments, where $0 \leq c_{ij} \leq 1$ and $X_m \in X_a$

Where: n = number of continuous stream segments in river network

l_i / l_j = quality weighted stream lengths of i^{th} / j^{th} segment

L_{TW} = quality weighted stream length of entire network

T = number of stream categories in river network

w_z = stream length quality weighting of z^{th} stream category

l_{iz} = length of i^{th} stream segment of z^{th} stream category

L_z = total length of z^{th} stream category in entire network

M = number of barriers passed between i^{th} and j^{th} segment

p_m^u / p_m^d = original upstream/downstream porosity of m^{th} barrier

Δp_m = change in porosity of m^{th} barrier due to modification

X_m = variable for m^{th} barrier between i^{th} and j^{th} segment

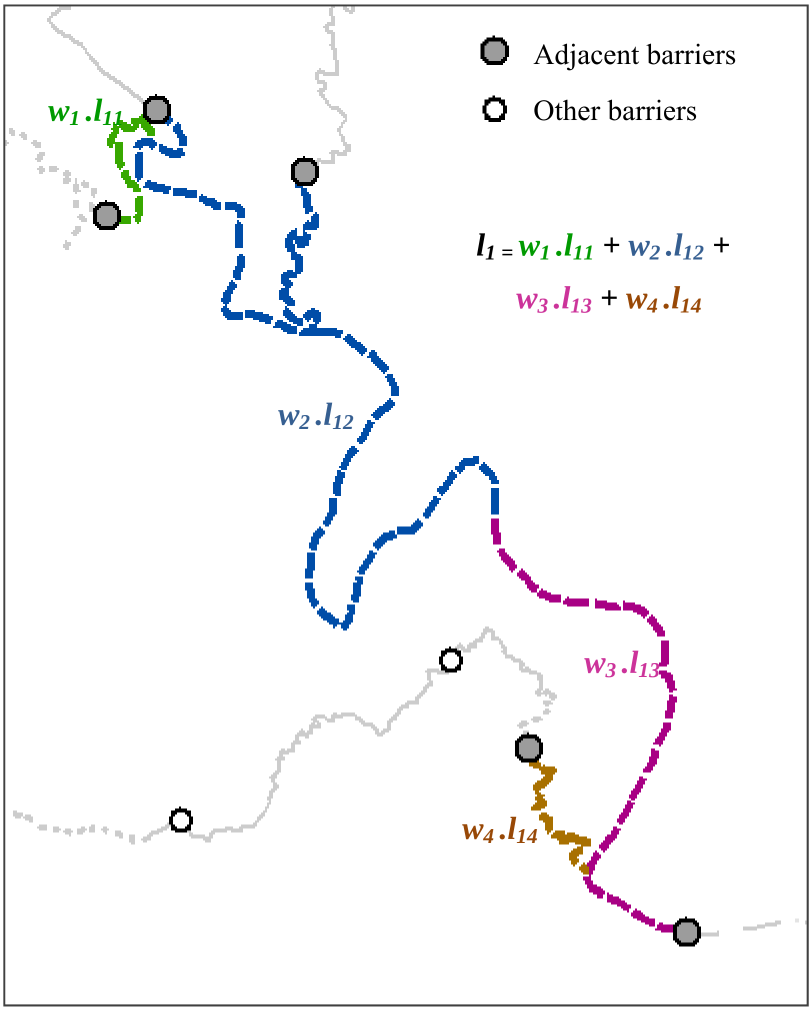


Figure 5.1 A continuous quality weighted stream segment made up of four stream sections of different length weighting. Total length of the segment is expressed through expanded form of Equation 5.4.

probability of movement between segments could be improved by increasing barrier porosity through physically modifying the barrier (Equation 5.6). Barrier porosity is a measure of how passable a barrier is either in terms of time or proportion of components which can pass through it, with porosity values ranging between zero and one (Kemp and O'Hanley, 2010). A fully impassable barrier has a value of zero and a fully passable barrier has a value of one.

Each barrier within a stream network was mathematically defined as an individual variable (X_a), thus the total number of variables solved for within the optimisation model was equal to the total number of barriers. A barrier could either be modified ($X_a = 1$) or left unchanged ($X_a = 0$), however no partial modifications could occur (Equation 5.7). Each barrier within the stream network was associated with a current state of barrier porosity and a potential future state of improved barrier porosity if the barrier was to be modified. A single solution of the connectivity restoration problem was composed of the unique combination of barriers which were selected to be either modified or left unchanged. The set of individual barrier porosities, and as a result the cumulative porosities, were adjusted in relation to the barrier modification solution and applied in calculating a new respective DCI value.

The optimal solution to the objective function was constrained such that the total cost of modifications of the selected barriers could not exceed the specified restoration budget (Equation 5.8). The budget constraint was adjusted into a simpler form if all barriers had an equal cost of modification (Equation 5.9).

The stream network connectivity restoration problem was solved with the use of a genetic algorithm function in *MATLAB R2011b* software package (Global Optimization Toolbox – Genetic Algorithm – Mixed Integer Optimization). The standard genetic algorithm in *MATLAB R2011b* was specifically calibrated for each stream network connectivity restoration problem, so as to reduce the time needed in the optimal barrier solution searching process. The output of the genetic algorithm was the maximum objective function value (DCI) and respective variable solution (combination of barriers to be modified) within the pre-defined constraints.

$$X_a \in \{0,1\}$$

Equation 5.7 Binary solution on modification decision of each barrier (X_a), where X_a is '1' if barrier 'a' is modified, and '0' if left unchanged ($1 \leq a \leq A$)

$$B \geq \sum_{a=1}^{A} C_a \,.\, X_a$$

Equation 5.8 Linear inequality constraint on optimisation solution, where total cost of solution (selected modifications) must not exceed budgetary limits

$$N \geq \sum_{a=1}^{A} X_a$$

Equation 5.9 Simplified form of linear inequality constraint, where all barriers are modified at an equal cost

Where: X_a = variable for respective a^{th} barrier

A = total number of barriers

C_a = cost of modification of a^{th} barrier

B = restoration budget

N = number of barrier modifications (applicable to optimisation problem where modification costs are equal for all barriers)

5.2.3 Data Sources and Variable Quantification

5.2.3.1 Introduction

Numerous stream network and barrier data are required as inputs to the optimisation model. This section describes the relevant data sources and the techniques in which the data were manipulated and applied to the optimisation model. First, a discussion of how segment length weighting values were calculated based on stream habitat quality of River Types is provided, followed by a description of the source of stream network data and the method of how it was used in the optimisation model. Finally, the method employed to quantify barrier modification character, namely barrier modification cost and final barrier porosity is outlined.

5.2.3.2 Stream Habitat Quality Weighting

Habitat quality within the Hunter River network was measured in relation to fish diversity found in each River Type described in Chapter 4. This approach is consistent with the concept that the physical habitat template dictates ecological pattern and process (Southwood, 1977). The Shannon-Wiener Diversity index was used to quantify habitat quality of each River Type. A quantitative electro-fishing survey of the Hunter River (1994 – 2010), supplied by NSW Industry and Investment was used to calculate habitat diversity values. The dataset contained information on the physical location of fish sampling, identified fish species and fish species abundance, in terms of total number of individuals and catch per unit effort (CPUE).

Sampling location data were imported into *ArcGIS 9.3* and sites were classified according to the underlying River Type. Accordingly, fish data across multiple sampling sites and times were arranged and grouped in respect to the underlying River Type of the sample site. Fish abundance for each River Type was quantified via CPUE data instead of abundance data to avoid bias associated with sampling effort. The diversity of each River Type was obtained with the Shannon-Wiener Diversity index, using total CPUE data for each fish species within the respective River Type (Equation 5.10). These River Type diversity values were utilised to weight habitat quality. The approach taken is similar to that recommended by Erős *et al.* (2011), where segment quality was weighted according to local fish species abundance.

$$H_z = -\sum_{i=1}^{S} p_i . \ln (p_i)$$

Equation 5.10 Shannon-Wiener Diversity index

Where: H_z = Shannon-Wiener Diversity index of z^{th} River Type

S = number of fish species in z^{th} River Type

p_i = proportion of total catch per unit effort belonging to i^{th} fish species in z^{th} River Type

5.2.3.3 Stream Network Data

Initially, fragmented stream network and barrier data were contained in *ArcGIS 9.3* as multiple shape files (Chapter 4). The stream network attributes required to calculate the DCI of a fragmented stream network were the physical segment character (quality weighted segment lengths) and spatial configuration between stream segments and barriers. This data was obtained with automated tools within *MATLAB R2011b*, which reconstructed the stream network data exported from *ArcGIS 9.3* into the necessary format for calculating the DCI. The outputs of the automated tools were in the form of two matrices. The first was populated with quality-weighted lengths of each segment, and the second contained the spatial configuration data of each segment, namely which segment and barrier were located directly downstream of each segment. The second matrix enabled the spatially explicit reconstruction of stream network topology with its respective barriers.

5.2.3.4 Barrier Modification Characteristics

Data obtained from the 'Bringing Back the Fish' project (NSW Industry and Investment, 2009) provided recent, detailed and publicly available information on the modification of barriers for improved passage of fish throughout NSW coastal stream networks. Data were reported for each restoration project, including the type of barrier that was to be modified, type of barrier modification implemented and the cost of modification. These data were used as input to the barrier characteristics component of the optimisation model, namely barrier modification cost and respective final barrier porosity. A summary of barrier modification options and an estimate cost is provided in Table 5.1. These barrier modification options are only representative of the projects implemented by NSW Industry and Investment (2009), and do not include all current available options, such as vertical-slot fishways, for which no recent cost information was provided. Even though the data did not include every available barrier modification option and were not necessarily wholly transferable to all barriers, it provided valuable insight into current network connectivity restoration procedures, patterns and costs implemented in NSW coastal catchments.

Cost estimates of barrier modification were obtained by first classifying the NSW Industry and Investment (2009) barrier modification data according to the barrier type being modified and the barrier modification option implemented in each restoration project. Each barrier type had been modified in a number of different ways, varying from project to project (Table 5.1), where an estimate cost was available for each project. This project cost data was used to calculate a mean cost of modification for each barrier modification option, with the data having already been classified and divided according to barrier type and barrier modification option. An average improvement in fish passage, also referred to as the 'barrier porosity', was associated with each modification option. No options for dam modification were available (Table 5.1), because no dams were modified as part of the 'Bringing Back the Fish' project. This exclusion of dams would have either none or a minimal influence on the optimisation analysis because of the exceptionally high costs associated with dam modification. The cost of implementing a fish lift on Tallowa Dam, Shoalhaven River, NSW, was broadly estimated at $26 million (NSW SCA, 2009), approximately a hundred times more costly than the second most expensive barrier modification option, the full-width rock-ramp fishway on a weir (Table 5.1), and approximately equal to the cost required to modify all the barriers in the entire Williams River network (Table 5.3).

Table 5.1 Types and average cost of restoration projects that were implemented in improving fish passage through barriers (Source: NSW Industry and Investment (2009)).

Original Barrier Type	Barrier Modification Options	Average Cost of Modification
Floodgate	Auto-tidal	$30,000
	Major Repairs	$60,000
	Minor Maintenance	$15,000
Dam	None	-
Weir	Full-width rock-ramp fishway	$300,000
	Repairs	$45,000
	Partial-width rock-ramp fishway	$175,000
	Removal	$150,000
Road Crossing	Box/Low Flow Culvert	$235,000
	Full-width rock-ramp fishway	$230,000
	Partial-width rock-ramp fishway	$130,000
	Removal	$15,000

5.2.4 Application of Optimisation Model

5.2.4.1 Introduction

The optimisation model was applied in two ways: first, to assess the level of sensitivity of connectivity restorations on model input data, and second to investigate the impact of optimal sequential barrier selection on decisions of barrier modification, and network connectivity. Both the sensitivity analysis and sequential optimisation were carried out on the Williams River, a major tributary of the Hunter River. The Williams River was chosen to test the optimisation model for several reasons. The region has a high ecological significance within the Hunter catchment, having substantial fish (van der Walt *et al.*, 2005) and mussel populations (Jones and Byrne, 2010), and all five geomorphic River Types were present in the stream network of the Williams River. The Williams River network contains 171 barriers, resulting in a comparably fast problem solving time, which allowed for a wide range of analyses to be carried out on the stream network. Three of the four barrier types were also found in this sub-catchment.

5.2.4.2 Sensitivity Analysis

A sensitivity analysis was carried out on four variations of the Williams River network, by varying the values of different attributes of the stream network and the barriers within it (Table 5.2). These were the 'reference' network, the 'habitat quality weighted' network, the 'variable barrier porosity' network and the 'variable barrier modification cost' network. The influence of each of the stream network and barrier attributes on the optimal level of stream network connectivity and the barriers selected for modification was identified with this procedure. Connectivity restorations were carried out over a range of budget values for each variation of the Williams River network (Table 5.3).

5.2.4.2.1 Stream Network and Barrier Model Assumptions

In the 'reference' network of the Williams River, stream lengths were left untouched and not weighted according to the underlying segment character or River Type. Each barrier in the network was characterised with the same values for barrier modification cost, initial barrier porosity and final barrier porosity (Table 5.2). This allowed for optimal barrier modification decisions to be based wholly on the physical structure and character of the network, without bias towards a specific barrier type. The initial state of barriers was assumed to be fully impassable, where each structure acted as a complete barrier to all components of the riverine landscape. Modification of each barrier was costed at a unit price of one and resulted in a barrier which was modified and fully passable (final porosity = 1). The cost of modifying all barriers and fully restoring network connectivity was $171 (Table 5.3 – Total network connectivity restoration cost), equal to the total number of barriers in the Williams River network.

The 'habitat quality weighted' network examined the influence of weighting according to habitat quality on barrier modification decisions and network connectivity. All stream segment lengths were weighted according to that associated with the Shannon-Wiener Index of the River Type of the respective stream segment. Similar to the reference network, each barrier was characterised with an equal barrier modification cost of one, initial barrier porosity of zero, final barrier porosity of one (Table 5.2) and a cost of $171 to fully restore network connectivity (Table 5.3).

The 'variable barrier porosity' network considered the influence of barrier porosity variables. Initial and final barrier porosities were randomly associated to each barrier with the use of *randi* function in *MATLAB R2011b* software. This function uses a discrete uniform distribution in random number selection. All initial barrier porosities were associated with random values less than 0.5, on the assumption that it would be highly unlikely to modify a barrier which was already more than 50 percent passable. Similarly, final barrier porosity values were randomly chosen to be equal to or greater than 0.5, based on the assumption that a barrier once modified would be at least 50 percent passable. Constraining barrier porosities (initial < 0.5, final ≥ 0.5) also mathematically ensured that barrier porosity always improved with modification. Random barrier porosity values were also constrained to increments of 0.1, so as to reduce computer processing time without comprising model output quality. Network segment lengths were not weighted according to underlying stream habitat quality and all barriers were assigned a unit modification cost of one. Again as with the previous two networks, budget required to fully restore network connectivity was equal to $171 (Table 5.3).

Stream segment lengths were not weighted according to habitat quality in the 'variable barrier modification cost' network. Here, each barrier type was associated with a set of possible modification options (Table 5.1), each of which was characterised with a unique modification cost and final barrier porosity. All barriers had an initial barrier porosity of zero. Each barrier was randomly assigned a modification option specific to the respective barrier type from Table 5.1, using *randi* function in *MATLAB R2011b* software package. The modification cost and final barrier porosity of each barrier was not equal, but rather associated to the respective randomly chosen modification option. Unlike with the first three networks, the budget required to modify all barriers and restore stream network connectivity to a maximum was $27.36 million rather than $171(Table 5.3). This value was equal to the total cost of modifying all barriers in the network, where barrier modification options and respective costs were randomly assigned according to Table 5.1.

Table 5.2 Stream network and barrier characteristics of the four variations of the Williams River network used in carrying out the sensitivity analysis.

	Network Characteristics			
Network Type	**Stream Quality Weighted**	**Barrier Modification Cost**	**Barrier Porosity**	
			Initial	***Final***
Reference	No	Equal ($1)	0	1
Habitat Quality Weighted	Yes	Equal ($1)	0	1
Variable Barrier Porosity	No	Equal ($1)	Random $(\geq 0, < 0.5)$	Random $(\geq 0.5, \leq 1)$
Variable Barrier Modification Cost	No	Randomised according to barrier type modification option	0	Random (associated to modification option)

5.2.4.2.2 Independent Optimisation Procedure

Decisions on barrier modifications for each of the Williams River network types in the sensitivity analysis were optimised for maximum network connectivity (DCI) over a range of budgetary values extending up to the total restoration cost of the entire network (Table 5.3). The optimal number of barriers for modification to maximise network connectivity, at each budgetary value, was independent of the optimal connectivity results at all preceding budgetary values. Barriers chosen for modification at a certain budget value might not necessarily form part of the optimal combination of barriers chosen for modification in the following budget value. Network connectivity optimisation was performed in relation to both the diadromous (Equation 5.3, $v = 1$) and potamodromous (Equation 5.3, $v = 0$) applications, for each variation of network type.

Optimal network connectivity and barrier composition was assessed at an interval of five percent of the total connectivity restoration cost, for the entire restoration cost range of each network type (Table 5.3 – ‘Coarse’). The first three network types had identical restoration budget attributes because all three networks had identical unit barrier modification costs (Table 5.3). Optimal network connectivity of the first five percent budget restoration interval was further evaluated at a finer restoration interval - 0.5 percent of total connectivity restoration cost. This provided detailed results on network connectivity restoration in the initial stages of the optimisation procedure. Optimisation data of each network was represented by plotting the restoration budget against its respective DCI value, referred to as the ‘Network connectivity-restoration budget’ curve.

Table 5.3 Total cost of barrier modifications and the respective restoration budget interval attributes used in the sensitivity analysis of each Williams River network type.

			Network Type			
			Reference	**Habitat Quality Weighted**	**Variable Barrier Porosity**	**Variable Barrier Modification Cost[1]**
Total Network Connectivity Restoration Cost			$171	$171	$171	$27360
Restoration Interval	**Fine**	Restoration Budget Interval	$1	$1	$1	$100
		Budget Range	$1 to $11	$1 to $11	$1 to $11	$100 to $1300
		No. of Restoration Intervals	$11	$11	$11	$13
	Coarse	Restoration Budget Interval	$10	$10	$10	$1350
		Budget Range	$11 to $161	$11 to $161	$11 to $161	$1350 to $27000
		No. of Restoration Intervals	$16	$16	$16	$20

[1] Unit: Thousands of dollars

5.2.4.3 *Sequential Optimisation*

A second optimisation procedure termed the ‘sequential optimisation’ procedure was implemented in the barrier selection process to maximise stream network connectivity restoration. The optimal set of barriers selected to restore stream network connectivity at one budget value might not necessarily be part of the optimal combination of barriers at a different budget value (O'Hanley, 2011). However, management authorities may restore connectivity of a stream network in numerous phases occurring over many years, rather than just a once-off restoration initiative for a specific catchment. The sequential optimisation procedure quantifies the impact that selecting optimal sets of barriers in phases (sequentially) has on the achieved levels of restored connectivity, considering that the optimal barriers modified in an earlier phase might not be optimal if and when further restorations can occur.

The sequential optimisation procedure was carried out using the generic optimisation model. Barriers were modified one at a time, selecting the single most optimal barrier for modification until all barriers in the network were modified. Each barrier selection was permanent and influenced all future restoration decisions and as a result, it ensured that all optimal restoration selections were made sequentially and not independently of optimal restorations at lower budget intervals. Once a barrier was selected as optimal for modification, its initial barrier porosity was adjusted to be equal to its final barrier porosity, which would make the selection permanent. In addition, this would result in the barrier not being selected for modification again, as its selection would not result in any improvements in network connectivity. The same stream network and barrier attributes were used as that of the 'reference' network in the sensitivity analysis. Each barrier was allocated a unit barrier modification cost, an initial barrier porosity of zero and final barrier porosity of one, and all River Types had the same habitat quality. This allowed for a direct comparison of both the stream network connectivity and barrier selections between the two independent and sequential optimisation procedures. Sequential optimisation of network connectivity restoration was performed in relation to both diadromous (Equation 5.3, $v = 1$) and potamodromous (Equation 5.3, $v = 0$) species behaviours. Similar to the independent optimisation procedure, data were presented with network connectivity – restoration budget curves.

5.2.5 Data Analysis

Data obtained from the sensitivity analysis were initially evaluated in terms of the individual diadromous and potamodromous species behaviour, after which the influence of choosing a certain index was assessed by a comparison of the two species behaviours. The sequential and 'reference' network optimisation datasets were compared to analyse the impact of optimisation procedure on level of connectivity improvement. Analysis of all the optimisation model datasets consisted of a comparison of 'Network Connectivity-Restoration Budget' curves and composition of selected barriers.

5.2.5.1 Network Connectivity-Restoration Budget Curves

Each network, of both diadromous and potamodromous species behaviours, from both sensitivity analysis and sequential optimisation datasets were characterised with a network connectivity-restoration budget curve. All datasets, with equal restoration budget values, were compared by calculating the differences in stream network connectivity. The influence of species behaviour on stream network connectivity was first assessed by calculating the difference between the two optimal stream network connectivity values that were achieved. Second, the potamodromous connectivity value of the optimal barrier selections for diadromous species behaviour, and vice versa, was evaluated. Both analyses examining the influence of species behaviour on connectivity were done for each variation of the Williams River network (Table 5.2).

5.2.5.2 Barrier Composition Analysis

The composition of selected barriers from the sensitivity analysis was analysed in three ways. First, the selected barriers were compared between the different variations of the Williams River network for each species behaviour and also between species behaviours. Second, barrier composition within each variation of the Williams River network was analysed for both species behaviours. Third, the level of importance of each barrier in restoring connectivity was assessed for both species behaviours. The difference in barrier composition between sequential and independent optimisation procedures was also assessed with the first type of analysis. Barrier composition of the sequentially selected barriers is implicitly nested, thus the second analysis was unnecessary on this dataset. In addition, sequential optimisation was only carried out on one variation of the Williams River network so as to identify the potential differences between procedures, thus also making the third analysis unnecessary. The methods employed to carry out the three types of analyses are described below.

The difference in composition of optimally selected barriers between two different Williams River network models was compared. For each budget restoration value, the number of selected barriers that were the same and the number that were different were identified. This analysis could only be performed between two Williams River network models that had equal restoration budget values.

Barrier composition was compared between different restoration budget values within each individual variation of the Williams River network models. This highlighted whether barrier selections were perfectly nested or had a more complex response. Analysis was carried out by assessing whether the set of selected barriers for modification at a lower budget value formed a sub-set of the selected barriers at the subsequent budget value. Comparison was performed for the entire range of tested budget values for each variation of stream network models. Each restoration budget value was compared to the lower adjacent budget value and the total number of barriers between the two budget values that were the same, newly added or newly 'removed' was identified.

The level of importance of each barrier in restoring Williams River network connectivity was assessed for both species behaviours. Barrier selection data of all four network variations were analysed simultaneously in order to rank the general level of modification importance of each barrier. The proportion of times that each barrier was selected for modification, out of all the coarse restoration budget intervals, was calculated for each barrier for all four network variations (Table 5.3). The data of all four network variations was amalgamated by calculating means and standard deviations for each barrier, on which the level of importance of each barrier was ranked. This procedure was performed separately for both diadromous and potamodromous species behaviours. Barriers were defined as 'critical' to restoring connectivity if they had been selected, on average at least 95 percent of times across all four variations of the Williams River network.

5.3 RESULTS

5.3.1 Introduction

The results of the optimisation model applied to the Williams River network are presented in two sections. The first section presents the stream network connectivity restoration results that were obtained from the sensitivity analysis. In addition, the composition of barriers selected to restore network connectivity is also presented. These sensitivity analysis results are given for the diadromous and potamodromous species behaviours individually and also as a combined solution. The second section of the results assesses the difference in the levels of improvement in stream network connectivity that is achieved between the sequential and independent optimisation procedures.

5.3.2 Sensitivity Analysis of Williams River Network

5.3.2.1 Optimising Restoration for Diadromous Species Behaviour

Connectivity curves for the four stream network types occurred in two response groups for diadromous species behaviour (Figure 5.2). The reference and habitat weighted networks had a very similar distribution with a maximum difference in connectivity of 1.66 units (Figure 5.2a). The second response group was made up of the variable barrier porosity and variable barrier cost networks. Connectivity between these two networks differed by a maximum of 6.7 units when 100 percent of barriers in both networks were modified (Figure 5.2a). However, the difference between the two connectivity response groups was significantly larger than within group variation. The maximum achievable network connectivity of the variable barrier porosity and cost networks was 14 and 7.3 units, which was substantially lower than the 100 units of the reference and habitat weighted networks.

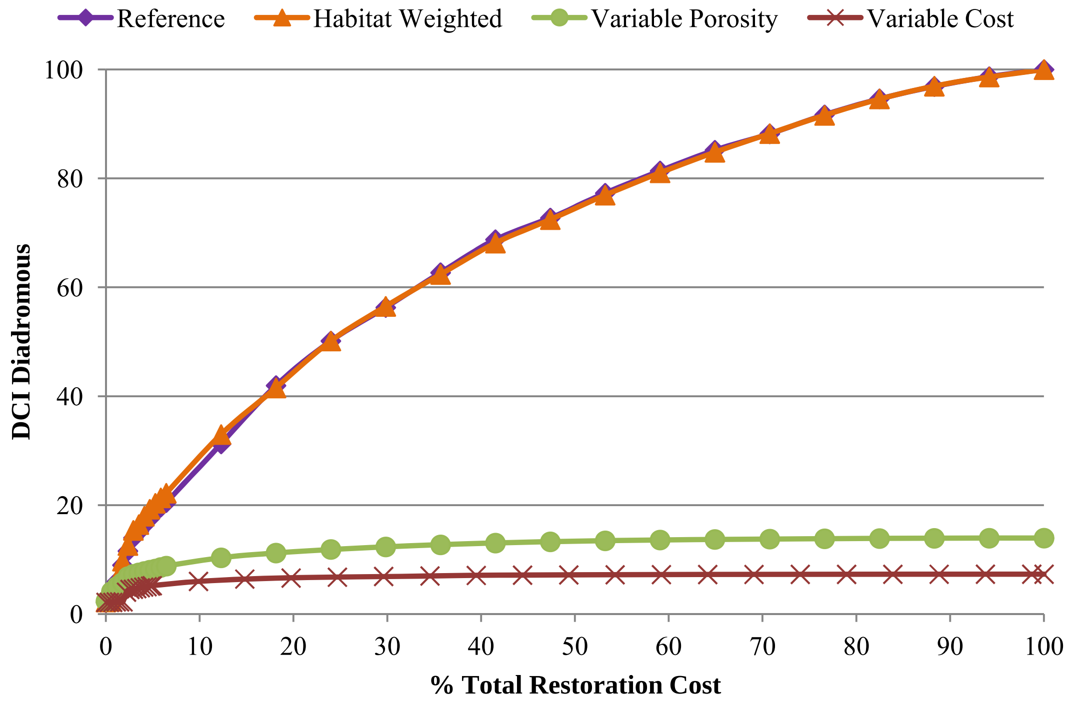
Connectivity curves for all four network types were broadly similar in shape, exhibiting the largest improvement in stream network connectivity with the modification of only a few barriers (Figure 5.2). Following these rapid early improvements, the rate of improvement in network connectivity gradually declined with an increase in the number of barriers modified. Approximately 15 percent of the reference and habitat weighted stream networks were opened with the modification of three percent of all barriers (Figure 5.2b). Modification of the final 40 percent of barriers only reconnected an additional 20 percent of both stream networks. The variable cost network had slow improvement in connectivity in the very early stages of restoration. Only 1.68 percent of total connectivity was restored with 1.83 percent of total restoration cost (Figure 5.2b), although the increase in connectivity of the variable barrier porosity and variable barrier modification cost networks thereafter was rapid; 76.3 and 82.8 percent of the total viable improvement in

DCI was achieved with 18.1 and 14.8 percent of the total restoration costs. Network connectivity increased by less than a percent of total DCI, for the final 24 and 36 percent of total restoration cost.

Once a barrier was selected as optimal for modification at a specific budget interval it was generally reselected to be modified at subsequent upper budgetary interval, irrespective of budget and stream network model assumptions (Figure 5.3). This pattern was evident at both the fine and coarse network connectivity restoration intervals, although it was more apparent at the fine interval where selected barrier composition only varied once for three of the four network types.

Variation in the composition of modified barriers was lowest for the variable barrier porosity network. Only two barriers were ‘removed’ between successive budgetary intervals, and barrier ‘removal’ only occurred for three of the 26 budgetary intervals (Figure 5.3c). In this instance, ‘removal’ refers to a barrier that was selected for modification at one budget interval but not selected at the following budget interval, thus being ‘removed’ from the set of selected barriers. Increased variability occurred within the habitat quality weighted network followed by the reference network, where the maximum number of barriers removed between adjacent budget intervals was five (Figure 5.3a and b). Composition of selected barriers between adjacent restoration decisions was dynamic within the variable cost network, for both the fine and coarse budgetary intervals (Figure 5.3d). The regular recurrence of barrier ‘removal’ between adjacent budget intervals (26 from 33) and the extent of change within a single budget interval (a maximum of seven barriers removed) was largest for the variable barrier modification cost network. However, the number of selected barriers that were the same was consistently more abundant than the number of barriers that were ‘removed’.

a. Actual improvement in stream network connectivity



b. Relative improvement in stream network connectivity

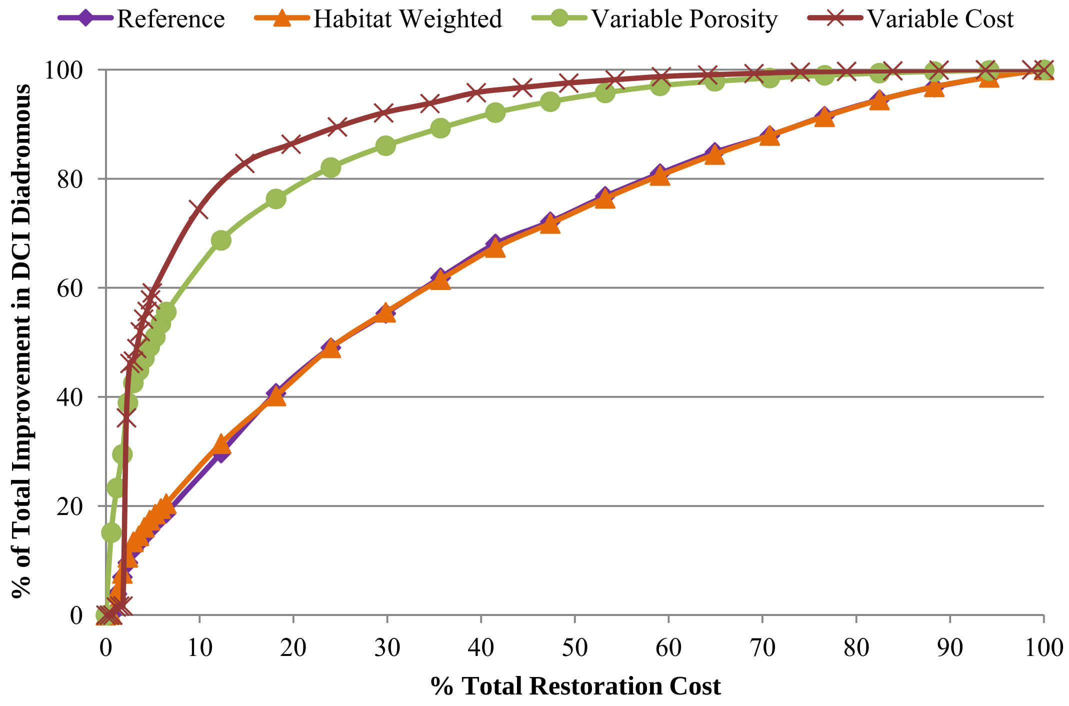


Figure 5.2 a). Actual, and b). Relative improvement in stream network connectivity (DCI) of the four variations of the Williams River network, as a function of percentage of the total network restoration cost, for diadromous species behaviour.

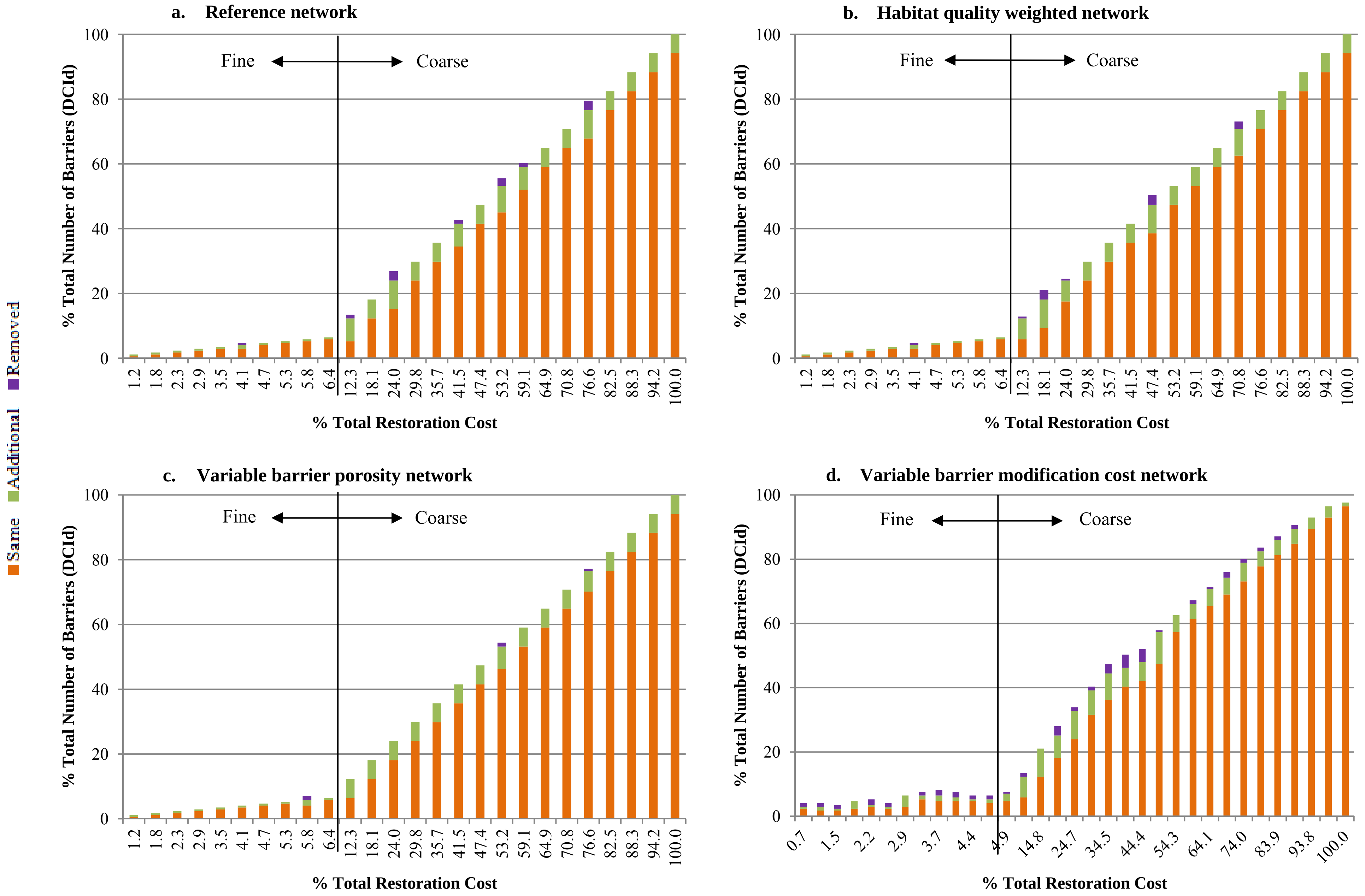
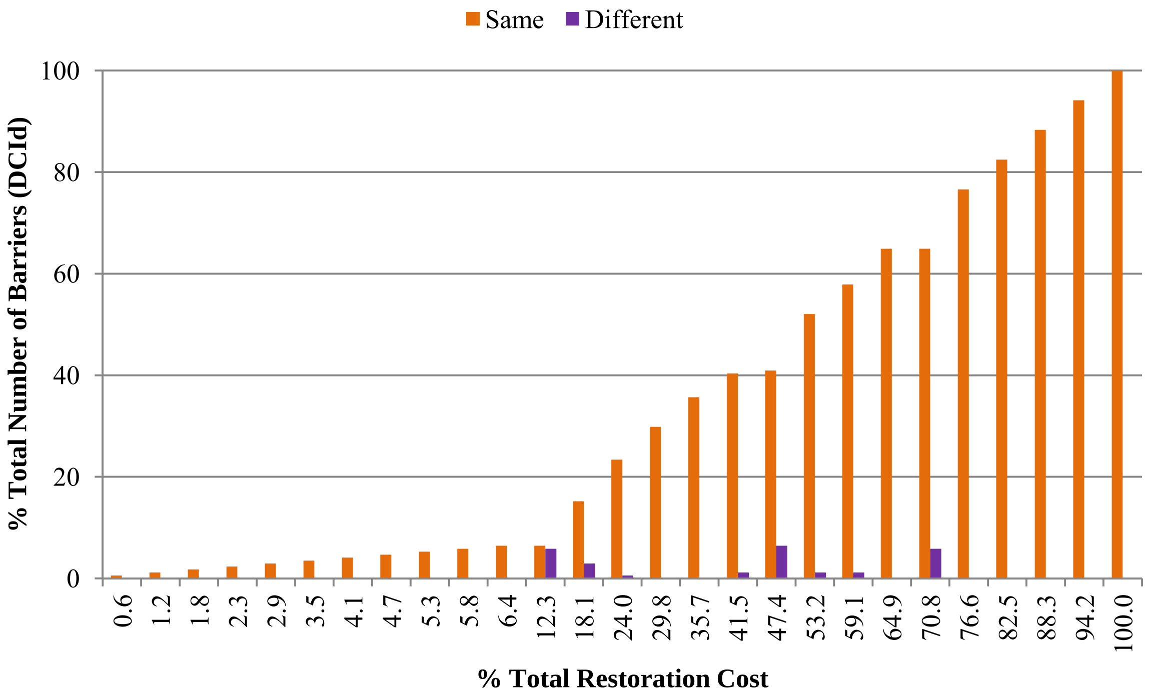


Figure 5.3 Within network comparison of the optimally selected barriers for restoring diadromous connectivity, of the a). Reference, b). Habitat quality weighted, c). Variable barrier porosity; and, d). Variable barrier modification cost networks. For each restoration interval, the proportion of selected barriers that are the same, the proportion that are newly selected and also newly removed, in comparison to the previous restoration interval are indicated.

Barriers selected for modification in the initial stages of the optimisation procedure were identical between the reference network and habitat quality weighted network – intervals less than 6.4 percent of total barriers (Figure 5.4a). Half of the coarse restoration intervals had some difference between selected barriers of these two network types. However, the number of selected barriers that were the same between the two networks was always greater than the number of differing barriers, for all budget intervals. The largest difference between barriers selected for modification of the two network types occurred at 47.4 percent of barriers modified, where six out of 81 barriers were identified to be different. The composition of selected barriers for the reference and variable barrier porosity networks was identical for the first four restoration intervals (Figure 5.4b). The number of differing barriers between the two networks gradually increased with an increase in restoration budget, to a maximum of 36 differing barriers, at 59.1 percent of total restoration cost. After this peak, the number of differing barriers steadily decreased until all selected barriers were the same. More of the selected barriers between the two network types were the same rather than different for majority of budget intervals. However, intervals in the range of 6.4 and 24 percent of total restoration cost were an exception, where a localised peak in the number of differing barriers occurred.

Nine barriers out of 171 were identified as critical in the diadromous network connectivity restoration process (refer to Appendix B for list of barriers). These nine barriers would reconnect 124.78 km or 18.64 percent of the Williams River network (Table 5.4). All barriers, except for one, were located in the downstream reaches of the main channel of the Williams River and they would open up 13.9 km of stream length with each barrier removed (Figure 5.5). The four barriers located furthest downstream on the main stem of the Williams River were consistently selected for modification, irrespective of budget interval and network type. Two barriers directly adjacent to the newly reconnected stream segments were selected in 99 percent of all budget intervals (Figure 5.5). The three barriers located furthest upstream on the Williams River were on average selected 95 percent of all budget intervals.

a. Habitat quality weighted network



b. Variable barrier porosity network

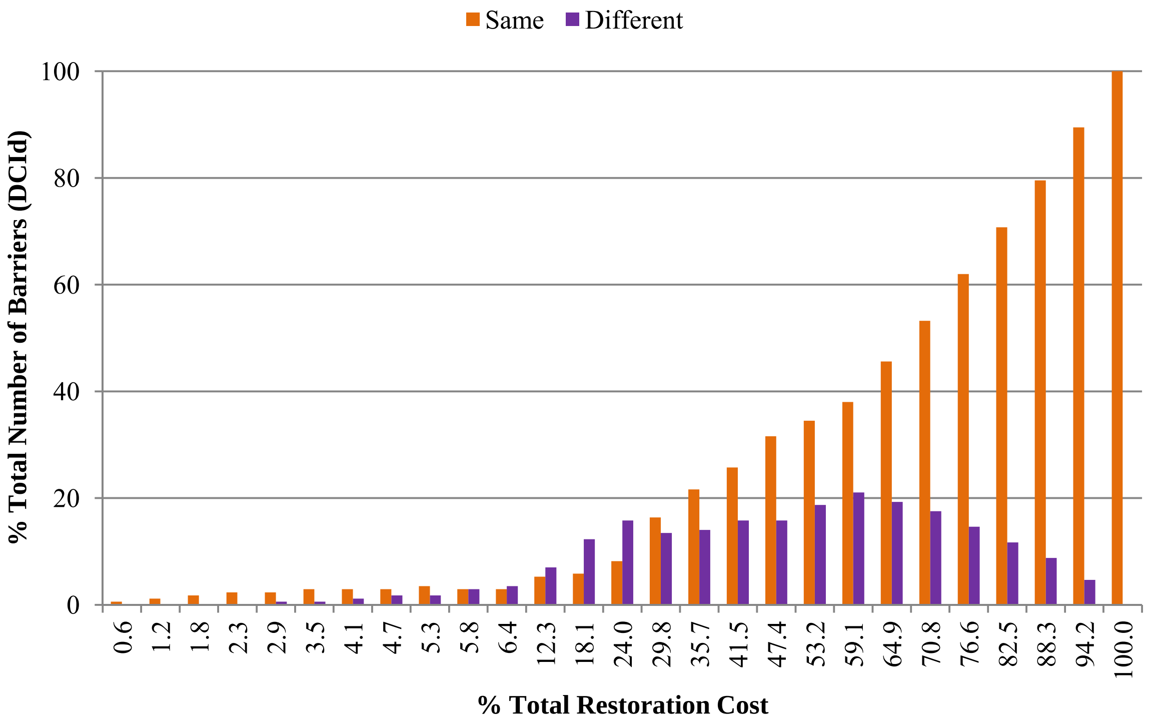


Figure 5.4 Comparison of optimally selected barriers for restoring diadromous connectivity between the reference network and a). Habitat quality weighted; and, b). Variable barrier porosity networks. The proportion of barriers that are the same and different between the networks types are indicated for each restoration interval.

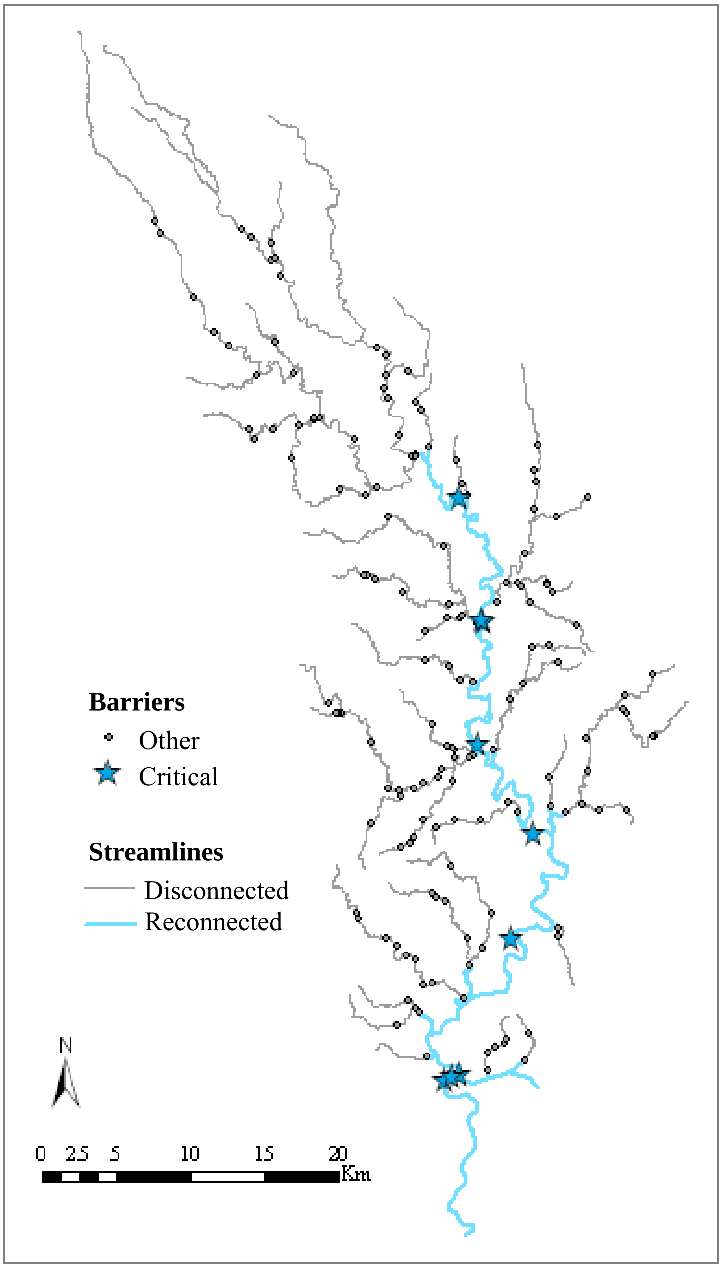


Figure 5.5 Spatial distribution of barriers which were consistently selected for restoration of diadromous network connectivity and the resulting reconnected segments.

Table 5.4 Reconnected Williams River network character as a result of modification of critical barriers for diadromous species behaviour.

Number of barriers	9
Reconnected stream length (km)	124.78
Average length per barrier (km)	13.86
Percent of total Williams stream length	18.64

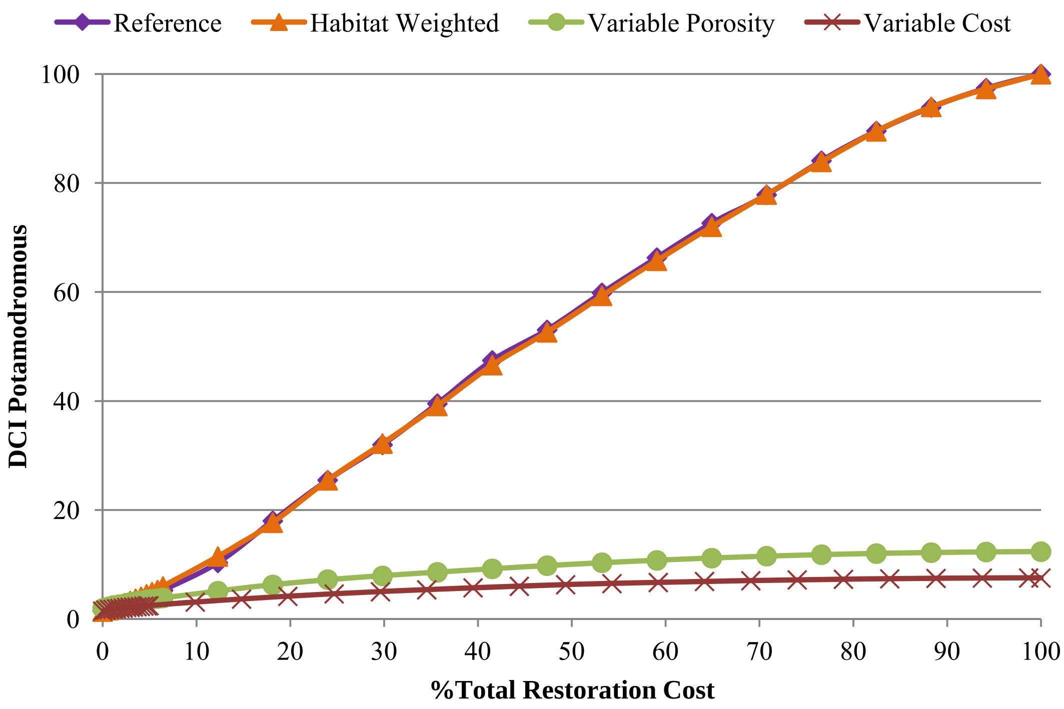
5.3.2.2 Optimising Restoration for Potamodromous Species Behaviour

Connectivity curves for potamodromous species behaviour of the four stream network types occurred in two general distributions (Figure 5.6). The reference and habitat weighted networks had connectivity curves of a similar nature as did those for the variable barrier porosity and variable barrier cost networks. The connectivity curves of the reference and habitat weighted networks were very similar, with a maximum difference in connectivity of 1.15 units at 12.3 percent of total restoration cost (Figure 5.6a). Modification of the first and last 18 percent of total barriers resulted in a slow improvement of 16.5 and 10.6 percent of total connectivity for both networks (Figure 5.6b). Otherwise, improvement in connectivity was linear. The variable barrier porosity and variable cost network types were similar, in that they improved network connectivity at a faster rate (Figure 5.6b). The majority of improvement occurred with the modification of the first few barriers. Modifying the first 24 percent of barriers achieved 51 percent of the total connectivity improvement. On the other hand, restoration of the last 46.8 and 50.7 percent of total network restoration cost only achieved an improvement of 20 percent of total connectivity for the variable barrier porosity and cost networks (Figure 5.6b).

The total improvement in connectivity was generally substantially greater for the reference and habitat weighted networks than the variable barrier porosity and cost networks (Figure 5.6a). Initial connectivity of the four network types before any restoration was similar – varying in the range of 1.38 and 1.96 units. However, at all other restoration budget intervals network connectivity of the reference and habitat weighted networks were significantly greater. Connectivity of the variable barrier porosity and cost networks improved by 10.41 and 6.07 units each, to a maximum value of 12.37 and 7.55 units. This was significantly less than the maximum achievable connectivity of 100 units for the reference and habitat weighted networks (Figure 5.6a).

Barriers selected as optimal for restoring potamodromous connectivity at a specific budget interval were generally reselected for modification at the subsequent upper budget interval (Figure 5.7). However, there was some variation in the composition of ‘removed’ barriers between the four network types, where barrier ‘removal’ has the same meaning as in Section 5.3.2.1. The variable barrier porosity network displayed limited variation between selected barrier compositions of adjacent restoration intervals (Figure 5.7c). Only five

a. Actual improvement in stream network connectivity



b. Relative improvement in stream network connectivity

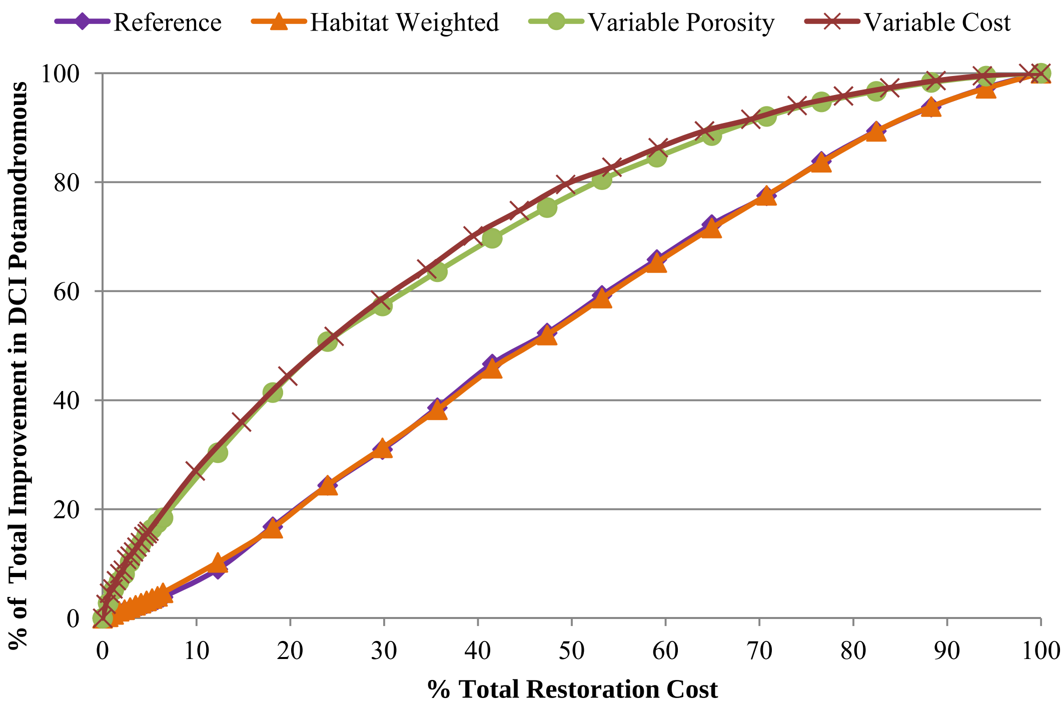


Figure 5.6 a). Actual, and b). Relative improvement in stream network connectivity (DCI) of the four variations of the Williams River network, as a function of percentage of the total network restoration cost, for potamodromous species behaviour.

restoration intervals or 20 percent of the total had selected barriers that were ‘removed’. Of the restoration intervals with ‘removed’ barriers, barrier composition differed by a maximum of two barriers at 64.9 percent of total restoration cost, with all other budget intervals separated by a single barrier. The reference and habitat quality weighted networks had a total of nine and seven restoration intervals with varying barrier composition, with a maximum of five ‘removed’ barriers (Figure 5.7a and b). The variable cost network had the greatest number of barriers which had been selected at a lower restoration interval and later ‘removed’ (Figure 5.7d). Three quarters of all optimised budget intervals had some level of variation in barrier composition. However, the number of similar barriers selected between budget intervals always exceeded the number of ‘removed’ barriers, except for at 1.5 percent of total restoration cost. Difference in barriers was larger at a fine restoration interval than the coarse restoration interval, where the maximum number of ‘removed’ barriers was nine in comparison to four.

Barriers selected for modification in the initial stages of the optimisation procedure were identical between the reference network and habitat quality weighted network – intervals less than 6.4 percent of total barriers (Figure 5.8a). The largest proportion of selected barriers that were different between the two networks occurred at 12.3 percent of total restoration cost. Differences in barrier selection between the two networks occurred for a further six budget intervals thereafter. The largest difference in barrier composition was six dissimilar barriers out of a total of 81 selected barriers – at 47.4 percent of total restoration cost.

Barrier composition of the reference and variable barrier porosity network types was similar when only a few barriers were modified (Figure 5.8b – 4.7 percent of total restoration cost and less). Selected barriers were identical for half of the restoration intervals or differed by a single dissimilar barrier for the other half. Similarity of barrier selections between the two networks diverged from restoration of 5.3 percent of total restoration cost. However, the number of selected barriers which were similar between the two networks exceeded the number of differing barriers for all but two restoration intervals (0.6 and 12.3 percent of total barriers). The number of differing barriers between the two networks stabilised from 12.3 percent of total restoration cost onwards, where at least ten barriers were different for 14 of the 16 restoration intervals. Five intervals had exactly 15 different barriers between the two networks, while only four exceeded the 20 barrier mark.

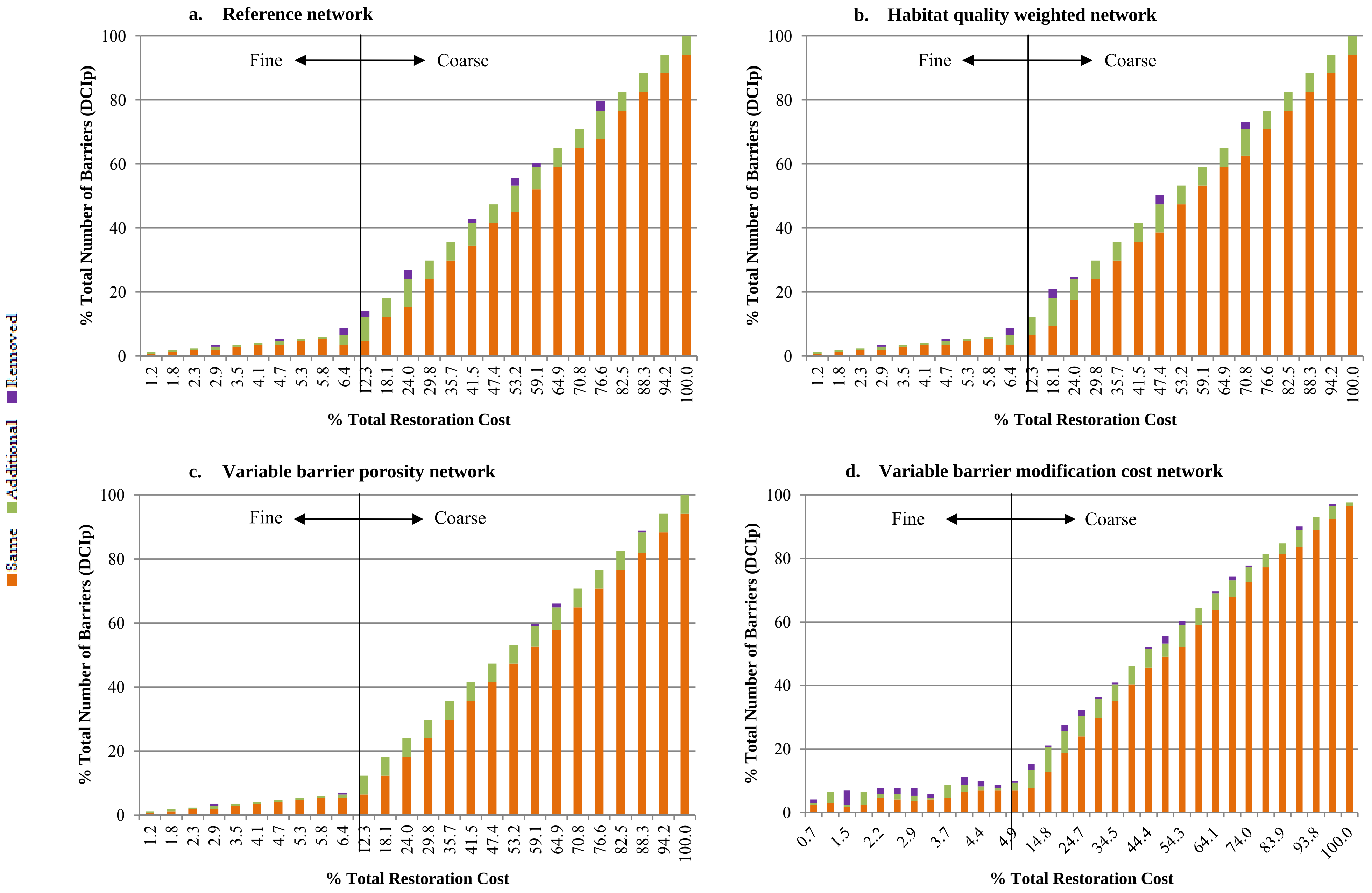
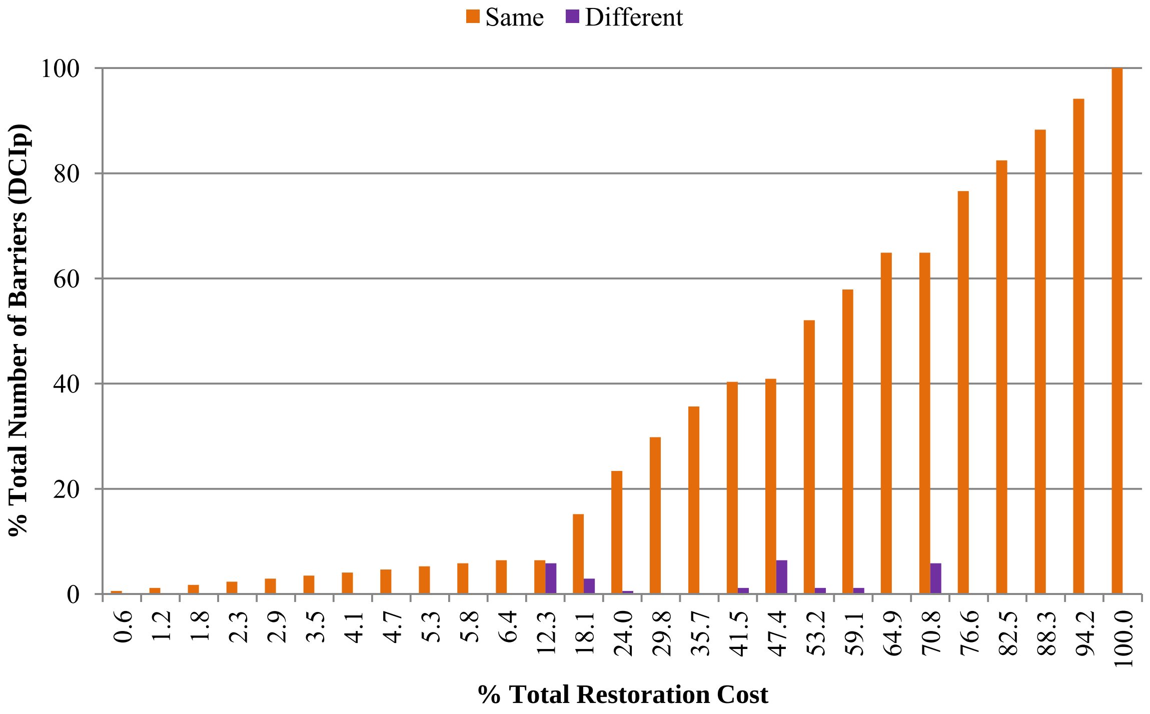


Figure 5.7 Within network comparison of the optimally selected barriers for restoring potamodromous connectivity, of the a). Reference, b). Habitat quality weighted, c). Variable barrier porosity; and, d). Variable barrier modification cost networks. For each restoration interval, the proportion of selected barriers that are the same, the proportion that are newly selected and also newly removed, in comparison to the previous restoration interval are indicated.

a. Habitat quality weighted network



b. Variable barrier porosity network

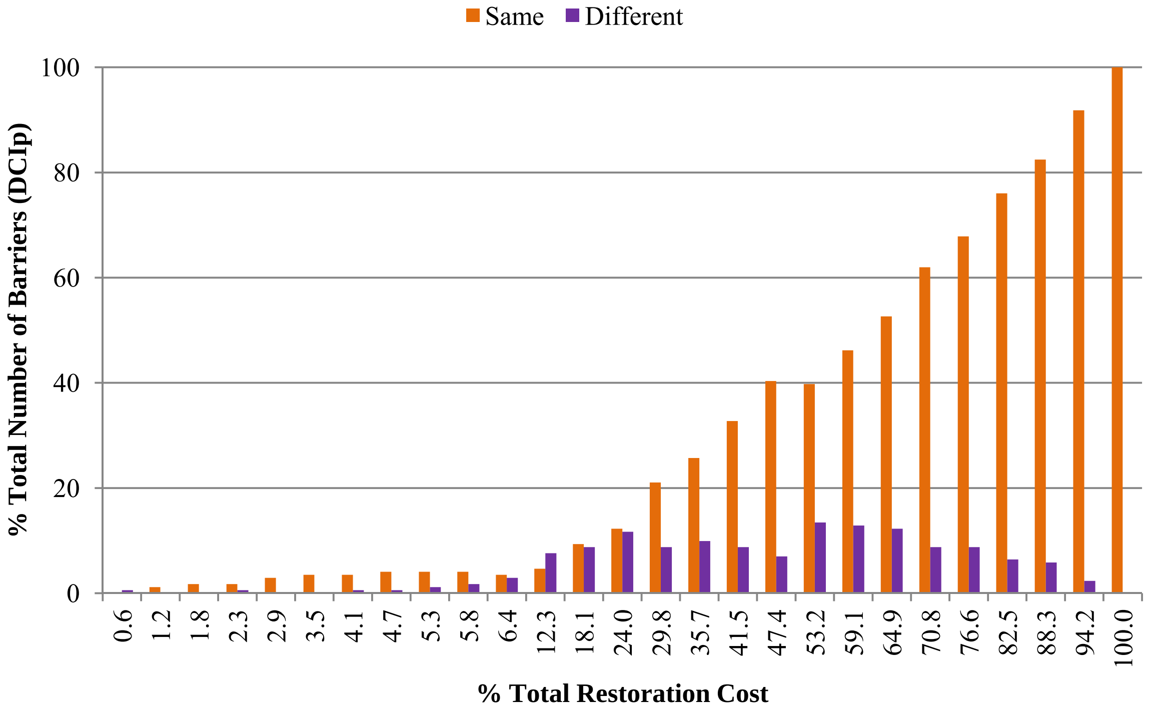


Figure 5.8 Comparison of optimally selected barriers for restoring potamodromous connectivity between the reference network and a). Habitat quality weighted; and, b). Variable barrier porosity networks. The proportion of barriers that are the same and different between the networks types are indicated for each restoration interval.

Eight structures were considered to be critical barriers for restoring potamodromous connectivity of the Williams River network (Figure 5.9; refer to Appendix B for list of barriers). These are located in the central region of the stream network of the Williams River (Figure 5.9). The three most downstream barriers were always selected to be modified, irrespective of the network type or restoration interval. Probability of selection decreased according to barrier location in an upstream direction. The following two upstream barriers were selected 99 percent of times, while the three barriers furthest upstream were selected, 96 percent of times. Each modified barrier, on average, reconnected 13.4 km of stream length, the total reconnection of 107 km representing 16 percent of the stream network (Table 5.5).

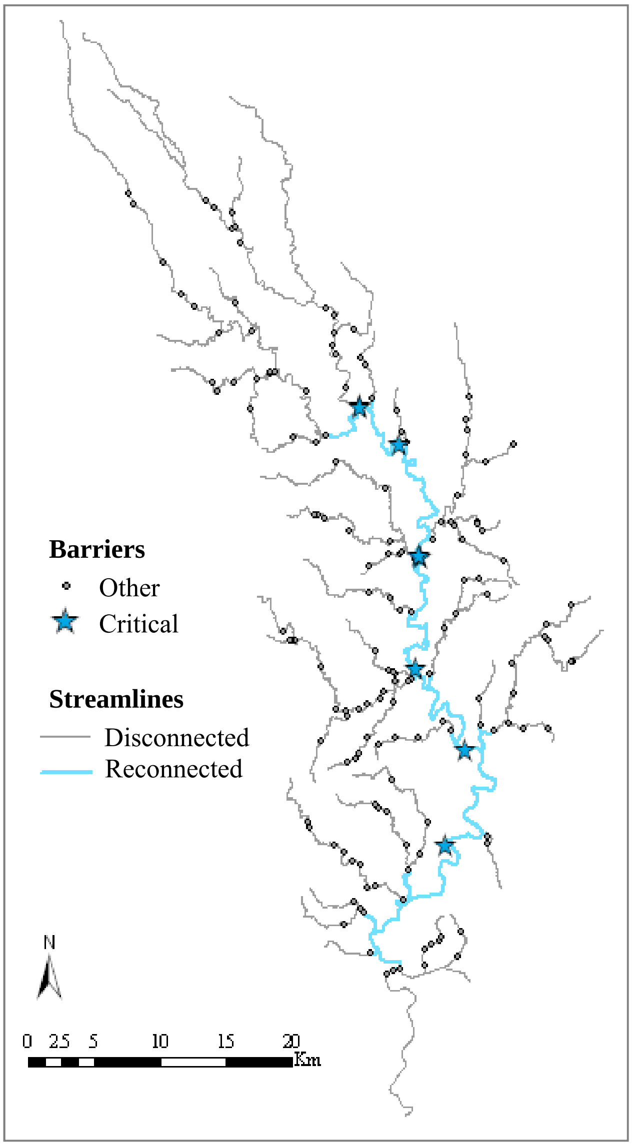


Figure 5.9 Spatial distribution of barriers which were consistently selected for restoration of potamodromous network connectivity and the resulting reconnected segments.

Table 5.5 Reconnected Williams River network character as a result of moficiation of critical barriers for potamodromous species behaviour.

Number of barriers	8
Reconnected stream length (km)	107.18
Average length per barrier (km)	13.40
Percent of total Williams stream length	16.01

5.3.2.3 Comparison of Diadromous and Potamodromous Species Behaviours

Optimal improvement in network connectivity was consistently more rapid for diadromous species behaviour than potamodromous, irrespective of network type (Figure 5.10). Difference in network connectivity between the species behaviours had the greatest improvement with the modification of the initial few barriers. Maximum difference was reached earliest for the variable barrier modification cost network, at 9.9 percent of total restoration cost, followed by the variable barrier porosity (12.3 percent), habitat quality and reference network (24 percent), listed in ascending order. Peaks in DCI units occurred at 2.89, 5.22, 24.69 and 24.67, or associated with 39, 35, 24.7 and 24.7 percent of the highest possible DCI of each network. After a maximum difference between connectivity, the two species behaviours converged at a slower pace, with the change in connectivity curves exhibiting a skewed normal distribution pattern. Change in connectivity between diadromous and potamodromous species behaviours, of the fully restored reference and habitat weighted networks, was zero (Figure 5.10a and b). However, the final diadromous network connectivity exceeded potamodromous connectivity by 1.6 units in the variable barrier porosity network. Potamodromous network connectivity of the fully restored variable barrier modification cost network was larger by 0.22 units than the diadromous connectivity.

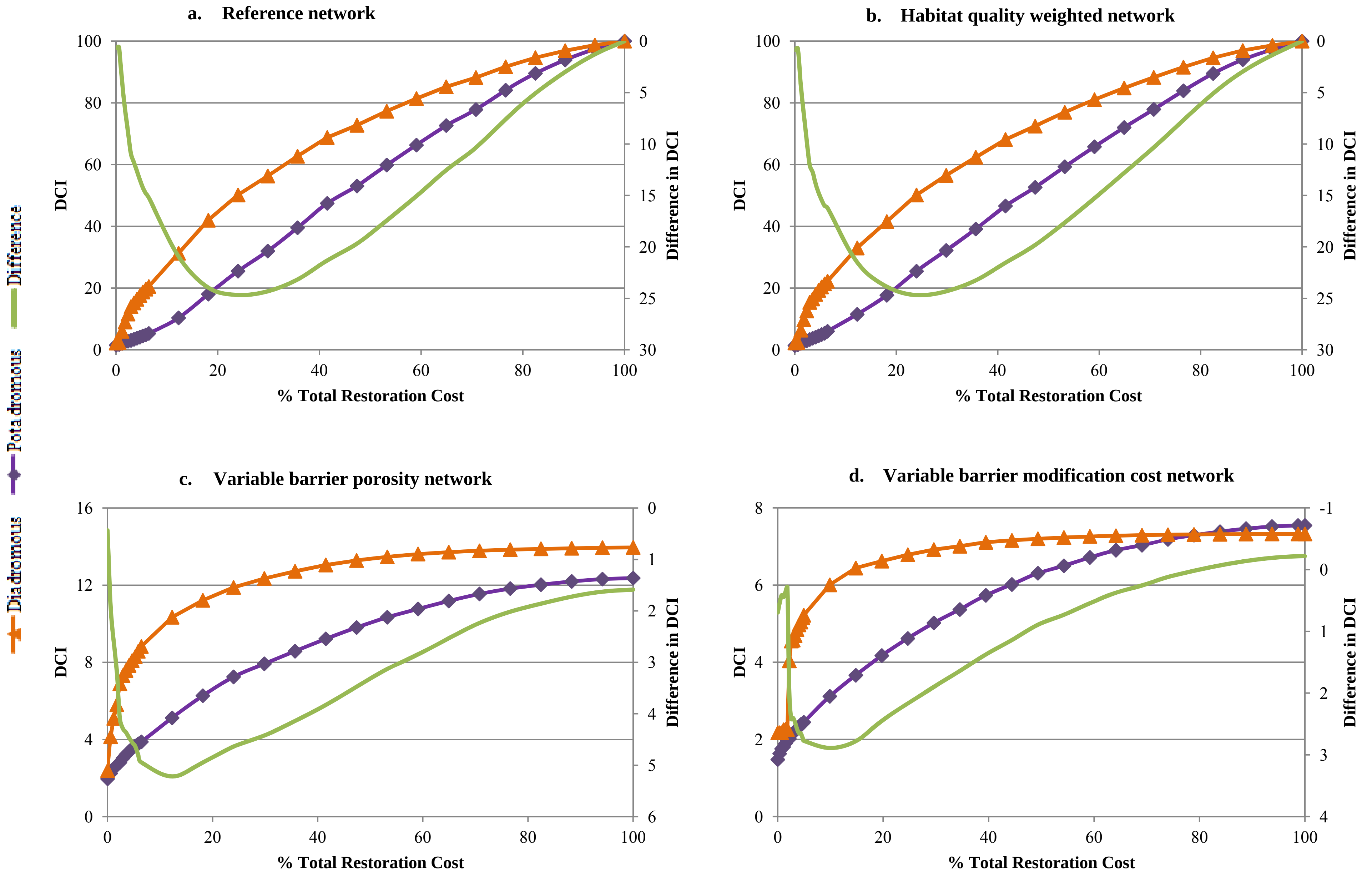
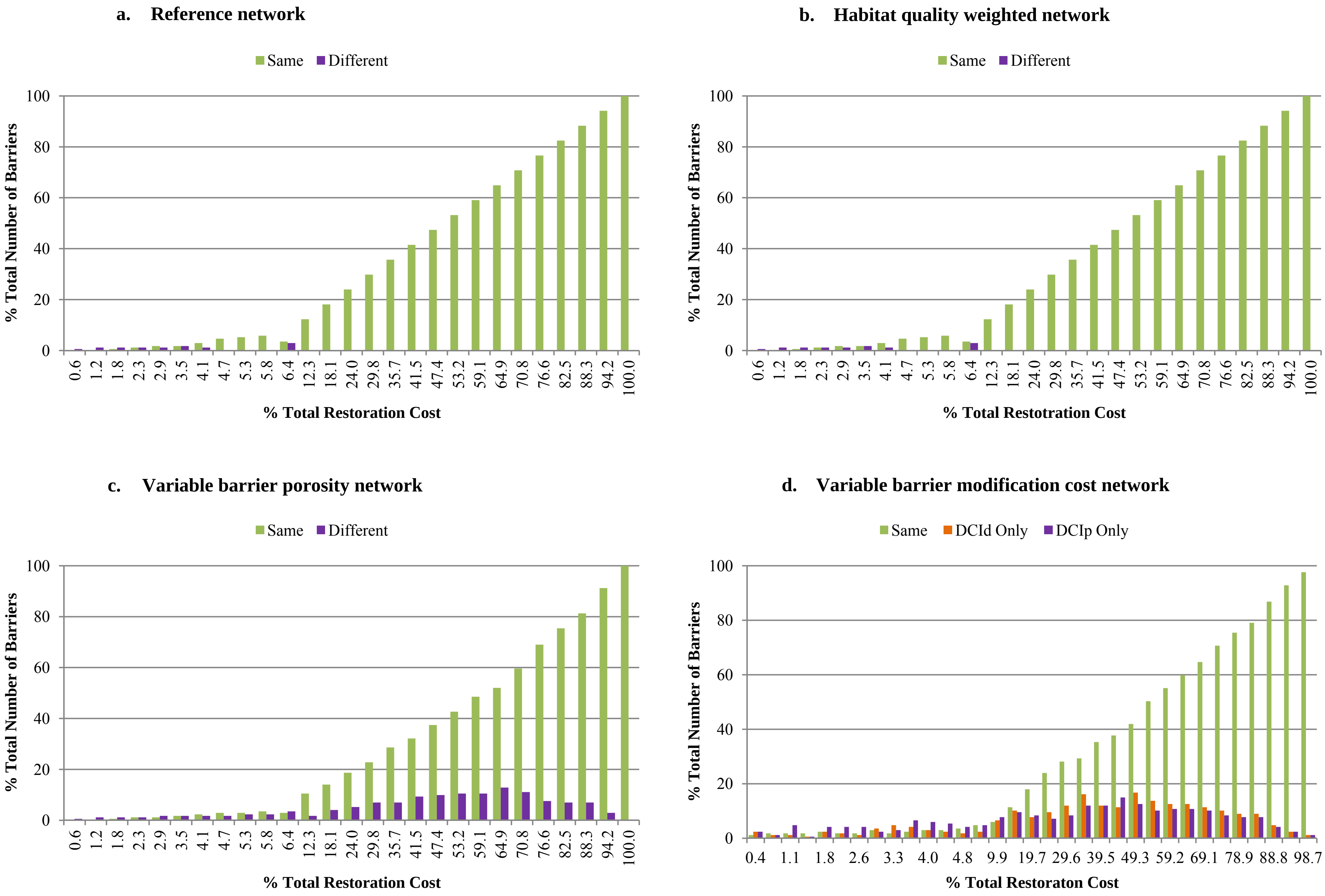


Figure 5.10 Difference in network connectivity (DCI) between diadromous and potamodromous species behaviours, as a function of percentage of the total restoration cost, for a). Reference, b). Habitat quality weighted, c). Variable barrier porosity; and, d). Variable barrier modification cost networks.

5.3.2.3.1 Barrier Composition and Spatial Variability between Species Behaviours

Barrier composition between the diadromous and potamodromous migration classes was similar for the reference network (Figure 5.11). The only differences occurred for eight restoration intervals in the early stages of optimisation, with up to 6.4 percent of total restoration cost (Figure 5.11a). Selected barriers between the two species behaviours were only dissimilar for the first two restoration intervals, after which barrier selections were the same for all intervals. The largest separation in the network occurred at 6.4 percent, where five of 11 barriers were identified as different. The reference and habitat weighted networks exhibited identical patterns in similarity of barrier composition between the species behaviours, for all restoration intervals (Figure 5.11a and b). Neither the variable barrier porosity nor cost networks had an identical set of barriers selected for modification between their diadromous and potamodromous species behaviours, irrespective of budget interval (Figure 5.11c and d). However, there was also an overlap in barrier selections between the two species behaviours for all restoration intervals, with the exception of the first two intervals of the variable barrier porosity network.

Comparison of the optimally selected barriers showed that there were more similar than different barrier selections between the two species behaviours, from ten percent of total restoration cost onwards (Figure 5.11). The maximum number of selected barriers that were different between species behaviours was 22 and 28 barriers at 64.9 and 49.3 percent of total restoration cost for the variable barrier porosity and cost networks. In the variable cost network, more barriers were selected when restoring for diadromous than potamodromous species behaviour, for restoration intervals greater than ten percent of total restoration cost (Figure 5.12). This occurred for ten of the 15 restoration intervals. In contrast, more barriers were selected to restore potamodromous connectivity for restoration intervals less than ten percent of total restoration cost. Potamodromous species behaviour had a higher count of selected barriers for ten of the 15 restoration intervals.



a. Reference network

b. Habitat quality weighted network

c. Variable barrier porosity network

d. Variable barrier modification cost network

Figure 5.11 Comparison of optimally selected barriers between diadromous and potamodromous species behaviours, for a). Reference, b). Habitat quality weighted, c). Variable barrier porosity; and, d). Variable barrier modification cost networks. Proportion of selected barriers that are the same and different between the two species behaviours are indicated for each restoration interval. In the variable barrier modification cost network, whether the 'different' barriers are only of diadromous or potamodromous species behaviour is indicated.

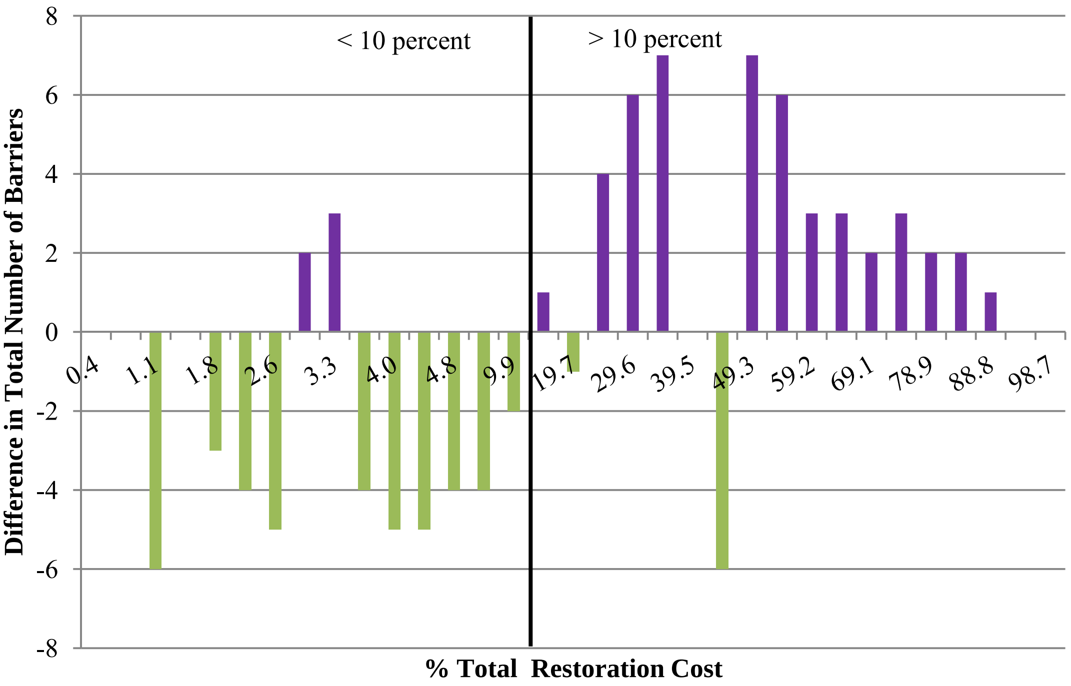
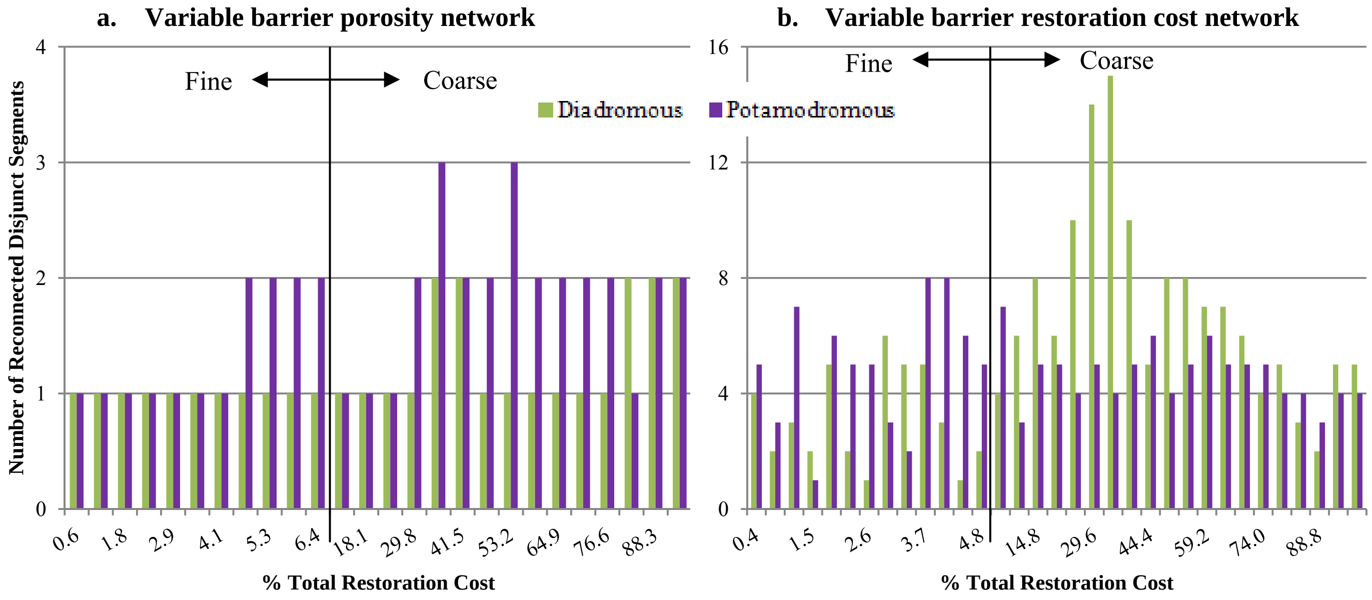


Figure 5.12 Difference in the total number of barriers restored between diadromous (positive) and potamodromous (negative) species behaviours of the variable barrier modification cost network.

Barriers were always selected for modification such that they would increase the size of a single continuous stream segment rather than multiple disconnected segments, irrespective of the restoration budget, in the reference and habitat weighted networks of both species behaviours. The variable barrier porosity network of diadromous and potamodromous species behaviours exhibited the same trend in barrier selection for 21 and 11 out of 26 restoration intervals respectively (Figure 5.13a). However, in certain instances, either two or three individual continuous segments were reconnected within a single restoration interval. This occurred when a barrier had a high initial porosity that minimally improved as a result of modification. Multiple individual reconnected segments were most abundant in coarser restoration intervals of potamodromous species behaviour. Out of a total of 15 intervals, nine were made up of two reconnected segments and two intervals were made up of three reconnected segments.



a. Variable barrier porosity network

b. Variable barrier restoration cost network

Figure 5.13 Number of reconnected disjunct segments of diadromous and potamodromous species behaviour for a). Variable barrier porosity; and, b). variable barrier modification cost networks.

Optimally selected barriers of the variable cost network resulted in the restoration of a single continuous segment only on three occasions, for both species behaviours (Figure 5.13b). The number of disjunct segments that were reconnected varied between one and six for diadromous migrations, and one and eight for potamodromous migrations, for restoration intervals less than 4.8 percent of total restoration cost. For 70 percent of these restoration intervals, potamodromous species behaviour had a larger number of disjunct segments reconnected. Diadromous connectivity had an increase in the number of reconnected disjunct segments for restoration intervals greater than 4.8 percent of total restoration cost. A maximum of 15 disjunct segments were reconnected at 34.5 percent of total restoration cost, while for potamodromous species behaviour the maximum number of reconnected disjunct segments was seven. Additionally, restoration resulted in either four or five disjunct segments being reconnected for 75 percent of the restoration intervals.

5.3.2.3.2 Implications of Index Choice on Optimal State

Difference between the optimal diadromous connectivity curves and the diadromous connectivity curves that were a result of the optimal potamodromous barrier sets varied slightly in shape between the four network types (Figure 5.14). Difference in diadromous network connectivity, as a result of the two optimal sets of barriers, had a positively skewed bimodal distribution and was greatest in the initial stages of optimisation for all network types. Maxima were located at 6.4, 6.4, 4.1 and 9.9 percent of total restoration cost and were characterised with differences of 18.3, 20, 5.2 and 3.8 units for the reference, habitat weighted, variable barrier porosity and variable barrier modification cost network types, respectively.

A difference in connectivity only existed in the first 6.4 percent of total restoration cost for both the reference and habitat weighted networks. Diadromous connectivity was equal from 12.3 percent of total restoration cost, irrespective of whether the diadromous or potamodromous optimal selected barrier sets were used (Figure 5.14a and b). After peaks in network connectivity, the difference in diadromous connectivity reduced rapidly to localised lows of 0.17 and 0.3 units at 12.3 and 19.7 percent of total restoration cost for the variable barrier porosity and cost networks (Figure 5.14c and d). This was followed by divergence in connectivity for both networks but to lesser extents. Local maxima of 0.64 and 0.26 units occurred at 41.5 and 49 percent of total restoration cost. Subsequently, difference in connectivity steadily decreased to 0.06 and 0.08 units, at 76.6 and 69 percent of total restoration cost, after which difference in connectivity decreased still further.

Difference between the optimal potamodromous connectivity curves and the potamodromous connectivity curves that were a result of optimal diadromous barrier sets were different for each individual network type (Figure 5.15). Potamodromous connectivity curves, as a function of both the respective diadromous and potamodromous optimal barrier sets, were alike for the reference and habitat weighted networks (Figure 5.15a and b). Network connectivity was identical between the two species behaviours from 12.3 percent of total restoration cost and onwards, for both network types. Peak differences in connectivity of 0.49 and 0.65 units both occurred at 1.75 percent of total restoration cost.

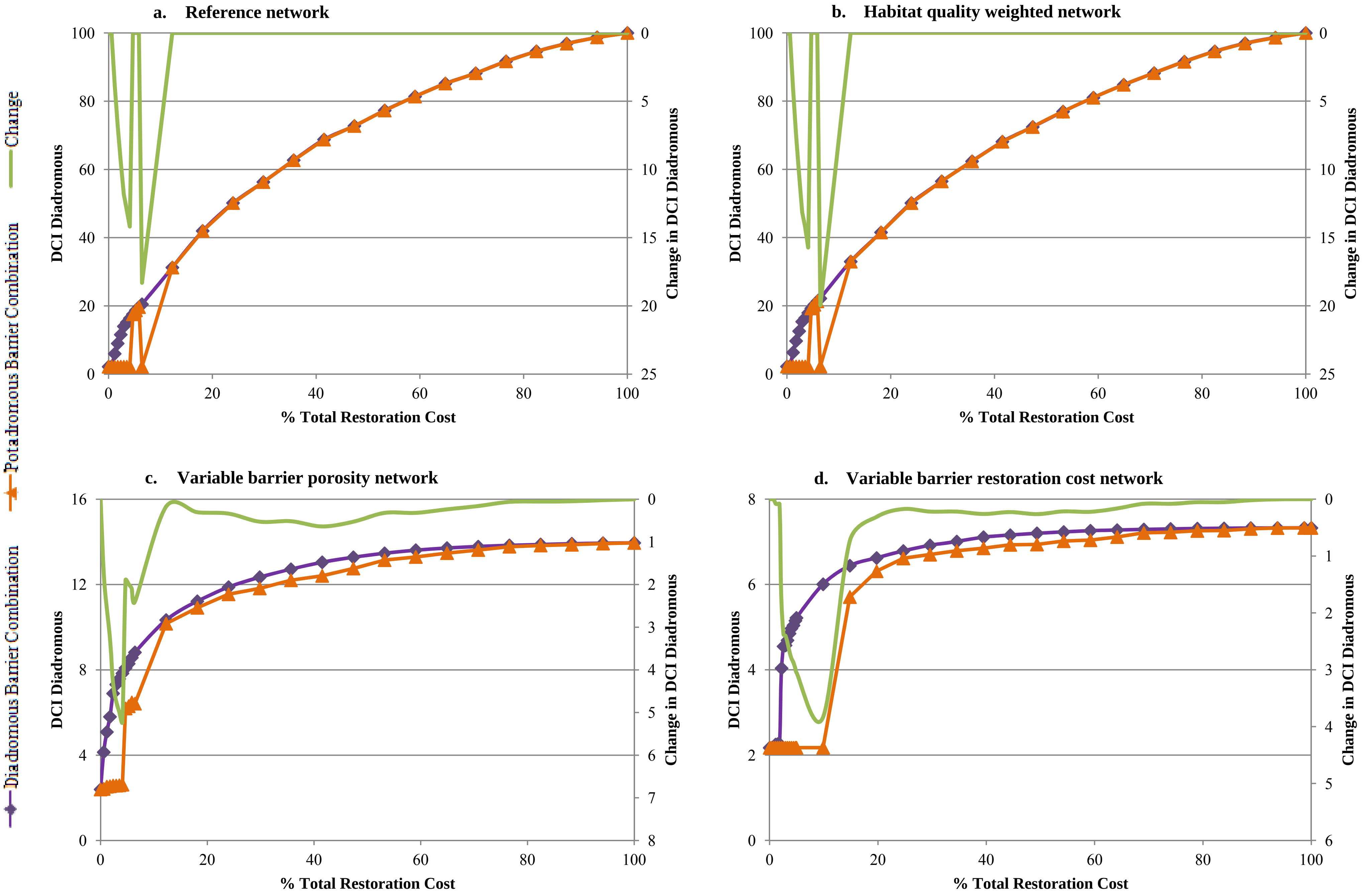


Figure 5.14 Difference in network connectivity (DCI) for diadromous species behaviour, of a). Reference, b). Habitat quality weighted, c). Variable barrier porosity; and, d). Variable barrier restoration cost networks. Change in DCI calculated using the optimal barrier sets of both diadromous and potamodromous species behaviours.

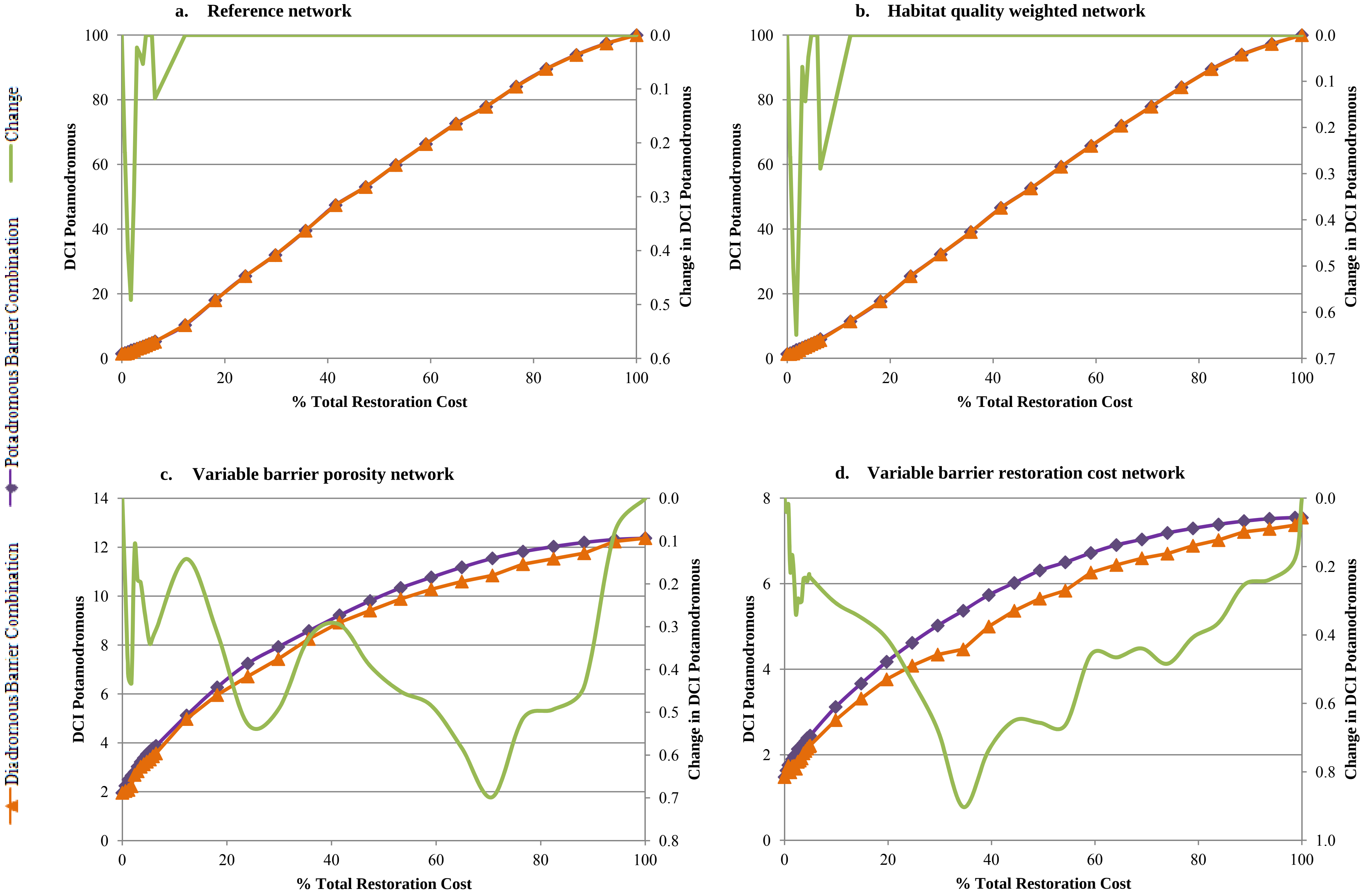


Figure 5.15 Difference in network connectivity (DCI) for potamodromous species behaviour, of a). Reference, b). Habitat quality weighted, c). Variable barrier porosity; and, d). Variable barrier restoration cost networks. Change in DCI calculated using the optimal barrier sets of both potamodromous and diadromous species behaviours.

The potamodromous connectivity curves that were a result of optimal diadromous barrier sets differed and were less than the optimal potamodromous curves, for both the variable barrier porosity and cost networks (Figure 5.15c and d). The difference between the two connectivity curves was greater for the variable modification cost network. Multiple local maxima and minima were recorded throughout both connectivity curves, especially for the variable porosity network. The smallest difference between the two connectivity curves occurred early and late in the optimisation procedure, for both network types. Maximum differences in connectivity of 0.7 and 0.9 units were located at 70.8 and 34.5 percent of total restoration cost, for the variable porosity and cost networks.

5.3.3 Sequential Optimisation

Network connectivity of diadromous species behaviour increased rapidly with the modification of a few barriers using the sequential optimisation process. Overall, modification of 3.5 percent of barriers reconnected 15 percent of the network (Figure 5.16a). The rate of improvement in connectivity decreased thereafter to approximately 0.7 units per barrier. However, a localised increase in rate of connectivity restoration occurred between 60 and 80 percent of total barriers modified. Network connectivity for potamodromous species behaviour improved at a low average rate of 0.52 units per modified barrier, for the initial 60 percent of barriers modified (Figure 5.16b). A rapid increase in connectivity from 31 to 67 units occurred in the range of 60 to 70 percent of total barriers modified, before the rate of improvement of network connectivity declined again.

The sequential optimisation procedure yielded different network connectivity distribution curves to those obtained using the independent optimisation procedure, in both species behaviours (Figure 5.16). With the modification of the first few barriers, network connectivity was equal between the two procedures for both species behaviours. Divergence in diadromous connectivity between the two procedures occurred from four percent of total barriers modified and three percent for potamodromous connectivity (Figure 5.16). Differences in restored network connectivity steadily increased for both species behaviours. Maximum differences in connectivity of 24.7 and 35.6 units were achieved when 41 and 59 percent of barriers were modified (Figure 5.18a and b; Figure 5.18c and d). Difference in diadromous network connectivity plateaued between 40 and 60 percent of modified barriers, after which the difference in connectivity declined. Difference in connectivity of potamodromous species behaviour abruptly decreased by 25 units, once the proportion of modified barriers increased from 60 to 70 percent. A secondary local maximum of 11 units occurred in the final restoration budget interval – 94 percent of barriers modified.

In the early phases of diadromous network connectivity restoration, i.e. less than 6.4 percent of barriers modified, the proportion of barriers selected that were the same was consistently more than the number of barriers selected that were different (Figure 5.17a). The number of differing barrier selections was larger than the similar selections in the restoration intervals between 12.3 and 41.5 percent of barriers modified. The latter coincided with the point of maximum difference in connectivity between the two procedures (Figure 5.16a). At this point, the majority of barriers selected for modification through sequential optimisation were located in the lower reaches of the Williams River network, while barrier selections as a result of the independent procedure extended further upstream (Figure 5.18). The largest number of different barrier selections occurred at 59.1 percent of barriers modified, after which the number decreased. The number of barriers selected by both procedures to be modified generally increased with an increase in the total number of modified barriers.

Barrier composition between the two optimisation procedures for potamodromous species behaviour exhibited similar trends to diadromous species behaviour (Figure 5.17). However, difference in selected barriers between the two procedures had a lower onset at 2.9 percent of modified barriers (Figure 5.17b). The number of different barriers exceeded the number of similar barriers for all restoration intervals in the range of 6.4 to 41.5 percent of barriers modified, at which a maximum of 38 differing barriers occurred. The proportion of differing barriers between the two procedures stabilised at an approximate 34 barriers between 35.7 and 59.1 percent of barriers modified, thereafter the proportion of differing barriers gradually decreased. Following this, the number of barriers selected by both procedures to be modified increased with an increase in the total number of modified barriers.

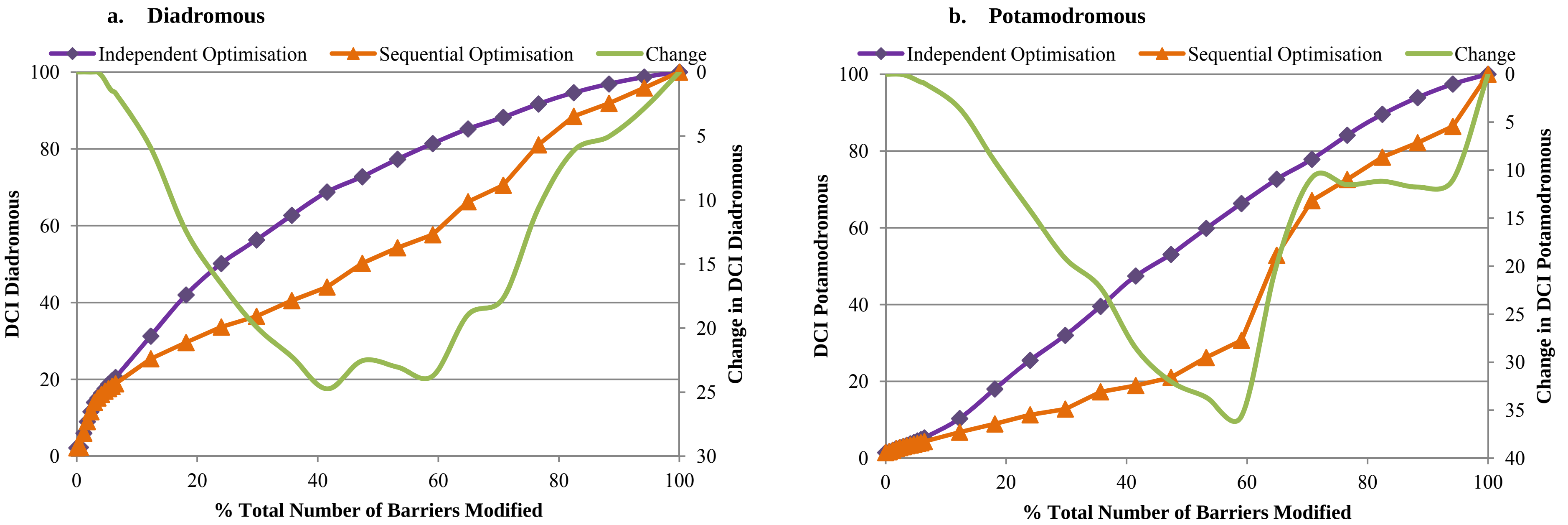


Figure 5.16 Difference in network connectivity between sequential and independent optimisation procedures for a). Diadromous; and, b). Potamodromous species behaviours.

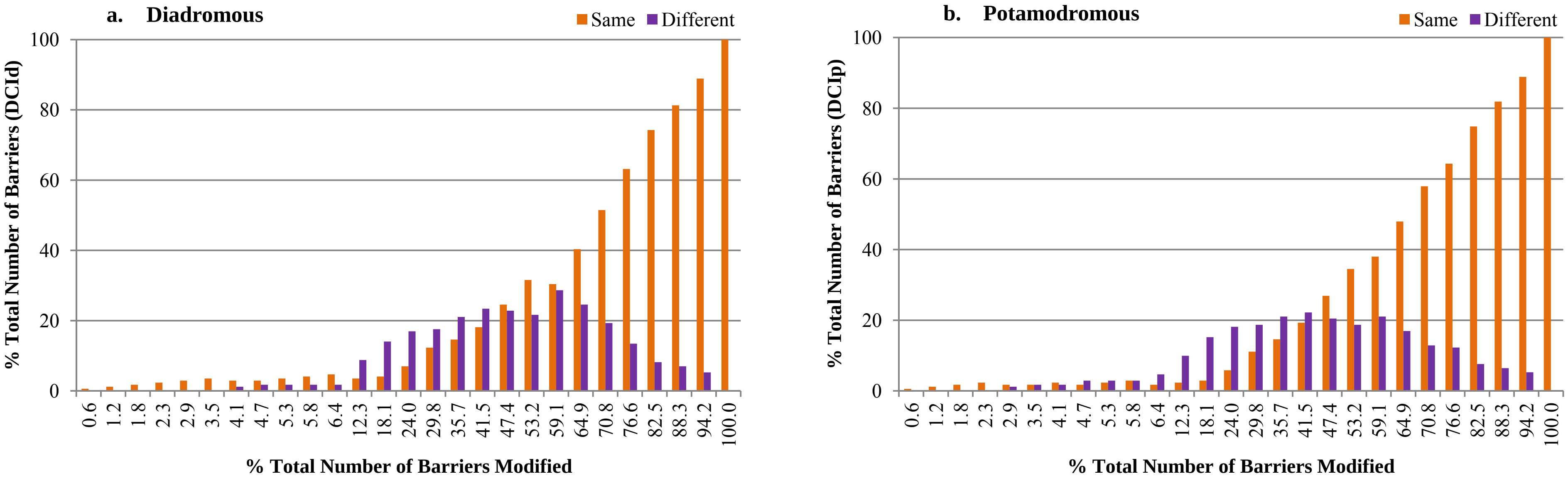


Figure 5.17 Difference in barrier composition from sequential and independent optimisation procedures for a). Diadromous; and, b). Potamodromous species behaviours. For each restoration interval, the total number of barriers which are the same and different between the two procedures are indicated.

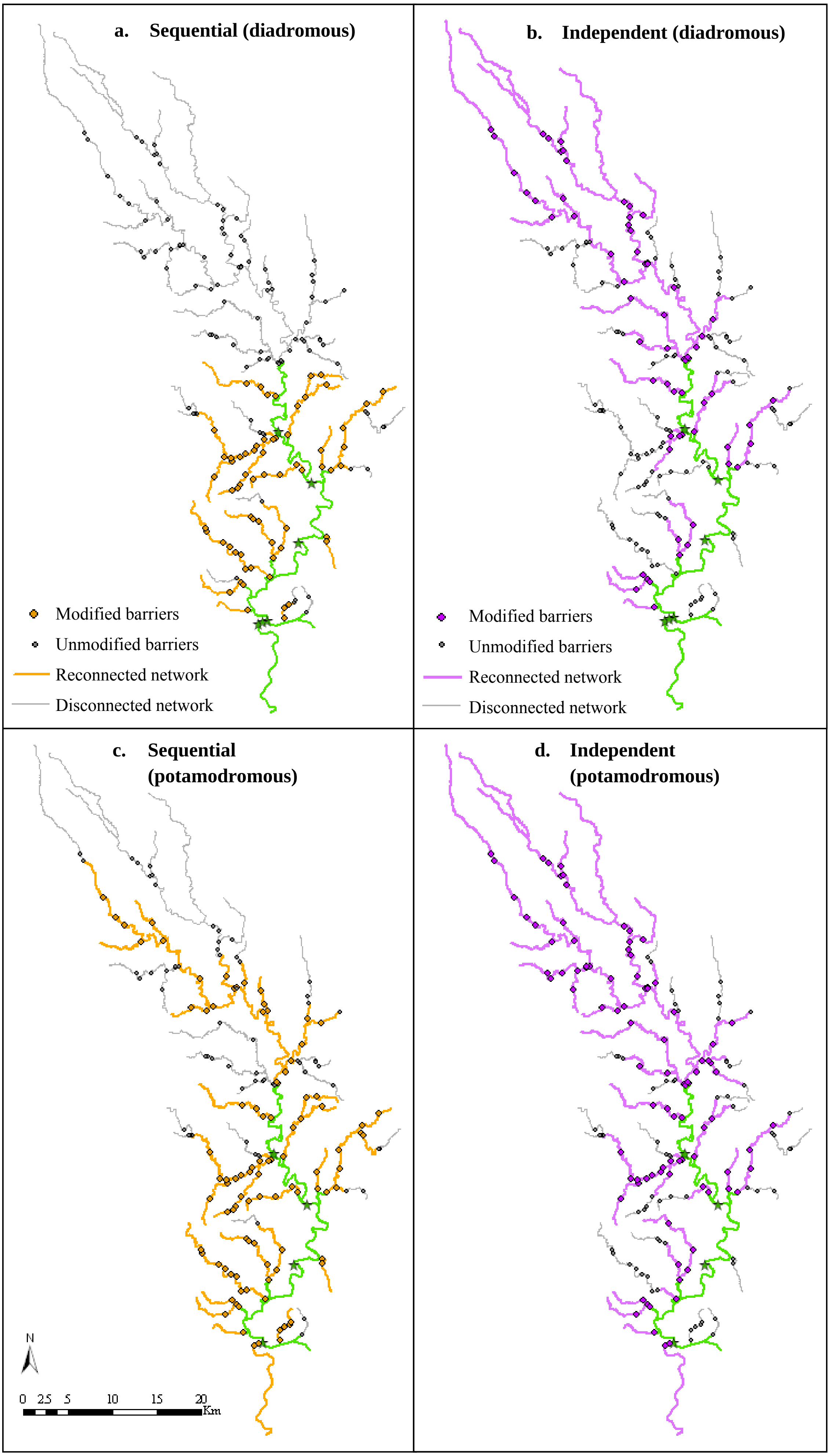


Figure 5.18 Composition of optimally selected barriers at point of maximum difference in network connectivity between the sequential and independent optimisation procedures. Difference in barrier composition between the two procedures for both diadromous (a. and b.) and potamodromous (c. and d.) species behaviour is indicated. Green streamlines and barriers indicate the last point at which both optimisation procedures selected the same barriers.

5.4 DISCUSSION

5.4.1 Variability in Stream Network Connectivity Restoration

The approach taken to optimally restore network connectivity within the Williams River demonstrated a distinct pattern between the number of barriers modified and the extent of improvement in connectivity. Overall, the rate of connectivity improvement was higher with initial barriers that were modified, then after some point the rate of improvement declined with further modified barrier (Figure 5.2; Figure 5.6). This pattern of restoration improvement is similar to that of Kuby *et al.* (2005) and O'Hanley and Tomberlin (2005), who reported large gains in habitat availability for anadromous species were achieved with either minor monetary investment or loss of socioeconomic services because of barrier modification via optimally targeting barriers. On the other hand, improvement in potamodromous connectivity of the reference and habitat weighted networks exhibited more of a linear pattern (Figure 5.6b). This is likely because of the quadratic form of the DCI potamodromous application equation (Equation 5.2). Distribution of connectivity improvement would have exhibited a similar pattern to the other networks if improvement in connectivity of these two networks was expressed in terms of the total length of the reconnected stream segment rather than the restored DCI of the network (O'Hanley, 2011). These findings have major implications for management, as substantial improvements in connectivity can be achieved with a limited restoration budget if barriers are optimally selected, thus ensuring maximum outcomes are achieved from limited monetary resources.

The results of this component of the thesis demonstrate that restoring connectivity in the Williams River was variable and sensitive to the nature of the stream network and barrier attributes. Connectivity was identified by Wiens (2002) being important to the integrity of riverine landscapes, and this was significantly influenced by patch quality, patch boundaries, patch context, organisms and scale. The first four of these attributes along with barrier modification cost were found to influence network connectivity in the Williams River. Restored connectivity varied for both species behaviours as a result of each attribute, but some attributes had more of an impact than others (Figure 5.2; Figure 5.6). In addition, the composition of barriers selected for optimally restoring connectivity was also variable (Figure 5.4; Figure 5.8). This suggests that certain stream network and barrier attributes might be more influential in connectivity restoration and in determining the composition of the optimal suite of barriers that are selected to maximise connectivity increases.

The quality of patches making up the Williams River network appeared to have a limited influence on connectivity restoration. The Williams River network is characterised by five geomorphically distinct River Types or patches, where each patch has a distinct weighting associated with habitat quality. The quality of patches was based on the assemblage diversity of fish species found in each River Type within the entire Hunter River network (see Section 5.2.3.2). The studies of Fleishman *et al.* (2002) and Palmer *et al.* (2000) have shown habitat quality to be the major driver of species abundance for certain species. However, the findings of this thesis suggest that habitat quality is not an important attribute when restoring connectivity, as in the Williams River patch quality had no influence. Patterns of restored connectivity were similar between the reference and habitat quality weighted stream networks for both species behaviours (Figure 5.2; Figure 5.6). In addition, the composition of selected barriers between the two network types did not vary (Figure 5.4; Figure 5.8).

This apparent lack of influence of patch quality on restoring connectivity in the Williams River could be a consequence of a number of factors. First, the range of habitat quality weightings was not large between the various River Types (Shannon-Wiener values in the range of 0.8 and 2.3). In this catchment, the most abundant River Types were highly weighted, while River Types with the lowest quality weighting were relatively rare. In addition, higher quality stream segments were located in the high priority connectivity regions of the network (central and downstream), while lower priority regions of the network (headwaters) were comprised of lower quality segments. This pattern of River Type quality is concurrent with the findings of Eros *et al.* (2011). Hence, these potential spatial associations may have further exacerbated the low impact of patch quality on connectivity restoration.

The use of assemblage diversity of fish species as an indicator of patch quality in the Williams River could have had a considerable influence on the connectivity restoration findings. In coastal south eastern Australian rivers, assemblage diversity of fish communities can have a positive correlation with distance along the stream network (Gehrke and Harris, 2000). The findings may have been substantially different if patch quality was based on a species whose habitat requirements had more distinct spatial associations, such as one associated with montane regions (Gehrke and Harris, 2000). In addition, the method applied in quantifying and associating biological presence to habitat quality can impose challenges. Commonly, the presence of biological species is considered a strong gauge of habitat quality, however an absence of species does not necessarily imply unsuitable or lower quality habitat (McDowall and Taylor, 2000). The distribution of fish, especially migratory types, within the Hunter River catchment may have been influenced by other factors in addition to habitat quality, such as restricted access to habitats due to the presence of artificial barriers and migratory behaviour. This might explain the lower diversity of fish assemblages found in habitat types located in the upstream regions of the stream network as to those located in the downstream regions. A similar approach to that proposed by McDowall and Taylor (2000) might be fruitful in addressing the influence of these various other factors on the distribution of migratory fish species, and thus habitat quality. Another possibility could be the use of less-migratory taxa such as macroinvertebrates, which might be a simpler and more applicable indicator of habitat quality. Irrespective of the method that is selected, it must be applied with caution and understanding, because other challenges have been encountered with associating biodiversity metrics to habitat quality (e.g. the influence of past land use (Harding *et al.*, 1998)). Thus, even though the findings of this thesis suggest that patch quality has negligible influence in restoring connectivity, its impact should not just automatically be ignored without further investigation and understanding.

Boundary characteristics, as denoted by the presence of barriers, did have a significant influence on connectivity in the Williams River. A boundary is characterised by its level of porosity, the extent to which a barrier inhibits ease of in-stream species movement past it (Kemp and O'Hanley, 2010). Different types of anthropogenic barriers have different porosities, thus impacting longitudinal connectivity of species to varying degrees (Brainwood *et al.*, 2008). A single barrier also affects different types of species, and even life stages, to different extents (Gehrke *et al.*, 2002; Holthe *et al.*, 2005) and barrier porosity can change with a change in flow conditions (Kemp and O'Hanley, 2010).

A reduction in the modified level of barrier porosity (patch boundary) in the Williams River was associated with a significant decrease in restored connectivity (Figure 5.2; Figure 5.6). Maximum connectivity of the variable barrier porosity network was only

13.96 and 12.37 units for diadromous and potamodromous species behaviour, in comparison to the 100 units of the reference network. Decreases in barrier porosity have been shown to considerably reduce the connectivity of a stream network (Cote *et al.*, 2009), by reducing ease of in-stream movement within the network. Additionally, selected barriers consistently differed between network types for both diadromous and potamodromous species behaviours, other than for the initial minor barrier modifications (Figure 5.4b; Figure 5.8b). A study carried out by Bourne *et al.* (2011) investigated the impact barrier porosity had on stream network connectivity and on barrier importance in restoring connectivity. Stream network connectivity and porosity of all barriers was individually quantified in relation to different type of species, life stage and barrier hydraulics. Bourne *et al.* (2011) found that even though network connectivity varied according to barrier porosity values for both species behaviours, the same barrier was still selected to be modified, suggesting that the priority of a barrier to be modified is not influenced by barrier porosity. The findings of this thesis (Figure 5.4b; Figure 5.8b) imply that the influence of barrier porosity on barrier selections is far greater and less predictable than that suggested by Bourne *et al.* (2011). These findings emphasise the importance of boundary characteristics and the extent of improvement in barrier porosity of individual structures when restoring stream network connectivity.

Restoring connectivity of the Williams River network was also influenced by the cost of individual barrier modifications. Improvements in connectivity increased in ‘steps’ with incremental increases in restoration budget, which was not evident for the other network types where all barriers had equal modification costs. These ‘stepped’ improvements in connectivity were notable for the small restoration budgets for the diadromous species behaviour (Figure 5.2). These thresholds appear to indicate ranges in budget for which near insignificant improvements in connectivity occur, as available funds are not adequate to target the most desirable barriers for restoring connectivity. Thus, near insignificant barriers which are within budgetary limits are being modified. This notion is further supported by the network having the most variable and least nested within network barrier composition (Figure 5.3d; Figure 5.7d). Slightly differing sets of barriers are consistently being selected without major stability in barrier selections over different budgetary intervals, unlike with the variable barrier porosity network (Figure 5.3c; Figure 5.7c). These results differ to the dam removal model of Kuby *et al.* (2005), where all dams selected for removal were perfectly nested. Kuby *et al.* (2005) selected dams according to tradeoffs between habitat gains and losses in socioeconomic services without accounting for removal costs in their model. This may explain the difference in results of the Kuby *et al.* (2005) study and this thesis. However, the findings of this thesis do concur with a study by O'Hanley (2011), which also found budgetary restrictions result in incremental increases in stream segment size and lack of perfect nestedness in barrier selections, especially at lower budget increments. Understanding the influence uneven barrier modification costs have on restoring connectivity is important, especially as it can prevent the unnecessary spending of funds on cheap barrier modifications which may barely improve stream network connectivity.

A series of 'critical barriers' were consistently selected as part of the target set of barriers for restoring connectivity in the Williams River network. These barriers were selected irrespective of the widespread variability both within (Figure 5.3; Figure 5.7) and across (Figure 5.4; Figure 5.8) network barrier selections when restoring connectivity for both species behaviours. Thus network topology (or patch context, Wiens, 2002) and the spatial configuration of barriers on the stream network may be overarching drivers in the restoration of connectivity. The results of optimal modelling showed that spatial configuration was more important in the early stages of restoring connectivity compared to other attributes such as patch quality, patch boundaries and restoration cost. These results concur with the recent studies by Fagan (2002); Benda *et al.* (2004); Ganio *et al.* (2005) and Campbell Grant *et al.* (2007) which all emphasise the importance of network topology of riverine landscapes and the potentially high influence topology can exert on pattern and process in these systems.

The majority of barriers critical to restore connectivity for both species behaviours were located on the Williams River itself. The Williams River includes some of the longest continuous stream segments in the network, and these form the backbone of restorations. A similar finding was reported by O'Hanley (2011). Furthermore, for diadromous species behaviour, critical barriers were located in the vicinity of the river estuary (Figure 5.5), while barriers were more central when restoring connectivity for potamodromous migrations (Figure 5.9). Even though no studies have examined the importance of specific barriers for optimally restoring stream network connectivity, the studies of Cote *et al.* (2009), Eros *et al.* (2011) and Rolls (2011) have reported similar spatial associations as found in this thesis. The association between barrier and stream segment location and the extent of influence they have on stream network connectivity, emphasises the close linkages between connectivity, network topology and barrier configuration.

The notion that there are ‘critical’ barriers for stream network connectivity has significant implications to the management of entire river systems. Firstly, substantial improvements in stream network connectivity can be achieved with a limited number of barrier modifications if the barriers are optimally targeted (Table 5.4; Table 5.5), because these barriers are associated with the highest possible improvements in connectivity. Secondly, high quality data on the stream network and barrier attributes might be unnecessary when modifying only a few barriers, as network topology appears to be more influential than other attributes when selecting the optimal set of barriers. This could reduce the amount of time needed for project planning, while simultaneously increasing the probability of selecting the correct barriers and achieving maximum outcomes without necessarily having the highest quality information with regard to the character of individual barriers.

5.4.2 Species Behaviour and Implications to Restoring Stream Network Connectivity

Species behaviour is an important factor influencing the restoration of stream network connectivity in the Williams River (Figure 5.10; Figure 5.11; Figure 5.14; Figure 5.15). Functional connectivity of landscapes is intimately linked to species type and its mobility characteristics, where the level of connectivity of a single landscape can differ between different species (Taylor *et al.*, 1993; Tischendorf and Fahrig, 2000). Bourne *et al.* (2011) showed that stream network connectivity was not only influenced by species type (salmon or brook trout) and life stage (individuals of varying size classes) but also by species behaviour. The findings of this thesis highlight the significance of species behaviour in connectivity restoration. Restored connectivity of the same stream network was shown to differ consistently for two species behaviours (diadromous and potamodromous) over the entire budget restoration range irrespective of both stream and barrier attributes (Figure 5.10). Species behaviour is important to the successful restoration of riverine landscapes and their respective ecosystems (Bond and Lake, 2003; Roni *et al.*, 2008) and individual migration types (e.g. anadromous and resident species) are being acknowledged and incorporated in connectivity restoration plans and efforts (Kuby *et al.*, 2005; O’Hanley and Tomberlin, 2005; Zheng *et al.*, 2009; O’Hanley, 2011). However, there have been few analyses that directly examine the influence of differing species behaviours on connectivity restoration. A notable exception is a study by Cote *et al.* (2009) which investigated stream network connectivity restoration for two life histories on the Big Brook River drainage network (Canada), fragmented by 15 anthropogenic barriers. Using a small data set Cote *et al.* (2009) showed restored connectivity differed for diadromous and potamodromous species behaviour, with the former improving at a more rapid rate; a finding similar to that of this thesis.

Species behaviour not only impacted connectivity restoration but also the composition of optimally selected barriers. Barrier composition consistently differed between the two life histories. The extent of variation was substantially smaller when there was no differentiation between barrier characteristics than when there was (Figure 5.11). Furthermore, the early stages in restoring connectivity (less than ten percent of total restoration cost) were marked by a divergence in barrier composition between species behaviour irrespective of either stream or barrier attributes. This is likely a consequence of the difference in critical barriers of each life history (Figure 5.5; Figure 5.9). Barriers closest to the river estuary were targeted first for species with diadromous species behaviour, while centrally located barriers were most critical for restoring potamodromous migrations. Similarly, Cote *et al.* (2009) identified that the most important barrier for restoring stream network connectivity was dependent upon species behaviour and differed accordingly. However, Bourne *et al.* (2011) found that species behaviour did not influence which barriers were selected to restore stream network connectivity; a single barrier was consistently prioritised as the most important to restore connectivity. Differences in these findings suggest that multiple attributes influence barrier importance when restoring for optimum stream network connectivity. Species behaviour can play a central role in selecting barriers in certain circumstances (Cote *et al.*, 2009), although these selections can be overridden by other dominant attributes such as stream network and barrier topology. The importance of various drivers in restoring connectivity of stream networks needs to be acknowledged in barrier prioritisation.

Different species behaviours are not equally sensitive to the extent of improvement when restoring stream network connectivity. The higher sensitivity of diadromous species to fragmentation has been illustrated through major declines in species abundance, extinctions and shifts in fish community structure from specialists towards generalists

(Reyes-Gavilán *et al.*, 1996; Joy and Death, 2001; Gehrke *et al.*, 2002; Guenther and Spacie, 2006). Similarly, the findings in this thesis suggest that restoring connectivity is also more sensitive for diadromous than potamodromous species behaviour, in relation to which barriers were selected to be modified. For example, when a suboptimal set of barriers were modified early in the optimisation procedure, improvement in diadromous stream network connectivity was nearly insignificant (Figure 5.14).

This difference in the restored levels of diadromous connectivity is potentially a consequence of the overriding influence that certain barriers exert. All diadromous species migrate between oceans and freshwaters regularly during certain phases of their life cycle (McDowall, 2007). Barriers located further downstream in the catchment are encountered more regularly by migrating species, and thus have a broader field of influence (Cote *et al.*, 2009; Rolls, 2011). Successful restoration of diadromous connectivity is highly dependent on first modifying barriers which are located furthest downstream before reconnecting the rest of the network. Otherwise, if upstream barriers are initially modified this will have negligible influence as downstream obstacles prevent any substantial catchment scale improvement in connectivity (Figure 5.14). This result concurs with other riverine restoration based studies, emphasising the importance of location at which restorations are carried out within the broader spatial context because of the influence other features in the catchment could have on restricting restoration success (Bond and Lake, 2003).

On the other hand, even when a sub-optimal set of barriers was selected to restore potamodromous stream network connectivity, there was a consistently smaller difference between the optimal and sub-optimal levels of restored connectivity (Figure 5.15). This lower sensitivity to barrier selection can be attributed to potamodromous species utilising habitats on either side of barriers, without the need for migrations to saltwater. However, potamodromous species behaviour should not be ignored in stream network connectivity restoration as fragmentation also exerts a formidable risk to the persistence of such species (Fagan *et al.*, 2002; Morita and Yamamoto, 2002).

Restoring stream network connectivity for the two species behaviours produced differing outcomes that may have substantial implications in managing Australian river systems. In New South Wales, connectivity and barrier modification projects are being implemented that target improvement in fish passage and aquatic habitats (NSW Industry and Investment, 2009). Contrary to the numerous connectivity restoration projects in the northern hemisphere which commonly aim to improve migration pathways for specific anadromous species (e.g. trout, salmon and lamprey) (Lucas *et al.*, 2009; Poplar-Jeffers *et al.*, 2009; Kemp and O'Hanley, 2010; Nunn and Cowx, 2012), restoring connectivity in Australian rivers has not necessarily been geared at a certain species or migration type but has rather targeted general improvements in ecosystem connectivity (NSW Industry and Investment, 2009). In these circumstances where no specific species is being targeted there is no clear cut species behaviour to restore to optimum levels.

The findings of this thesis suggest that if a limited number of barriers are to be modified this should be done in respect to diadromous species behaviour, due to the significantly higher sensitivity towards the initial barrier selections (Figure 5.14; Figure 5.15). Such selections should ensure that the most evident migration barriers with the largest scale of influence are modified first, without substantially compromising the potential improvement in connectivity for potamodromous species. However, once the priority downstream barriers are modified, further improvement in diadromous connectivity substantially wanes, while improvements in potamodromous connectivity become more sensitive to barrier selections (Figure 5.12; Figure 5.14; Figure 5.15). These results suggest that if a larger number of restorations are to be carried out, stream network connectivity should be restored for potamodromous species behaviour. Implementing the approach developed in this thesis could aid management of entire riverine landscapes and their aquatic communities, by acknowledging and incorporating multiple species and their behaviours in connectivity restoration plans.

5.4.3 Restoring Stream Network Connectivity Sequentially

Numerous techniques have been developed to prioritise barriers for restoring stream and habitat connectivity. For example, Poplar-Jeffers *et al.* (2009) ranked barriers primarily according to their level of impassability and then according to the upstream habitat availability, while Kemp and O’Hanley (2010) elaborate on a common technique where barriers are prioritised in relation to a ratio quantifying upstream habitat gain against cost of restoration. Other methods incorporate additional variables into barrier prioritisation, such as fish stock levels in the surrounding streams or spatial configuration of barriers in relation to each other and accessibility to the river estuary (e.g. NSW DPI, 2006b; Nunn and Cowx, 2012). These types of techniques generally score all barriers part of an inventory once off and accordingly prioritise their order of modification. The benefits of such methods are that they can be easily implemented and adapted to vaguely measure various ecological criteria. However, the study of O'Hanley and Tomberlin (2005) which was reworked by Kemp and O'Hanley (2010) found restoration outcomes of score-rank techniques to be significantly sub-optimum in comparison to optimisation procedures.

The optimisation approach implemented in this thesis had numerous strengths in comparison to traditional score-and-rank techniques (Kemp and O'Hanley, 2010). Commonly, score-and-rank techniques assess barriers independently, not accounting for the spatial configuration of barriers within a stream network and non-additive effects of multiple modifications. On the other hand, the optimisation approach consistently incorporates the spatial configuration of the entire river system in question, guaranteeing that maximum improvements in stream network connectivity are consistently achieved. In addition, the approach can be applied to restore connectivity for different objectives, such as to accommodate two different species behaviours. In formulating the problem, clearly defined objectives in terms of both barrier and riverine landscape attributes are required which ensure that there are high levels of transparency in the barrier selection process (Kemp and O'Hanley, 2010). This is not necessarily true with other techniques which might be susceptible to higher levels of subjectivity, especially in ranking the importance of certain structural and ecological attributes to connectivity restoration.

Sequentially selecting barriers for modification generally resulted in sub-optimal improvements of stream network connectivity in the Williams River. Connectivity was only optimally restored early in the optimisation procedure when a few barriers were modified, after which the subpar differences steadily increased before decreasing again. These sub-optimal trends were evident for both types of species behaviour (Figure 5.16) and appear to be partly driven by a lack of perfect nestedness between barrier selections at differing budget intervals (Figure 5.3a; Figure 5.7a). In the independent optimisation procedure, a certain barrier might have been selected as optimal at one level of restoration, while at a different level it was not necessarily part of the respective optimal combination. However, the sequential procedure is implicitly nested, with all previous modifications forming a subset of future decisions, leading to the onset of sub-optimal connectivity and divergence between barrier compositions. Commencement of divergence coincided with the first break from perfect barrier nestedness (Figure 5.3a; Figure 5.7a; Figure 5.17).

Divergence from optimal connectivity by the sequential procedure could have been partly driven by selecting barriers at such a fine interval (a single barrier at a time). For example, a long and continuous length of the Williams River network was fragmented by two barriers which were in close proximity to each other. By sequentially restoring diadromous connectivity a barrier at a time, the potential of reconnecting a long stream segment through the simultaneous modification of two barriers was overlooked due to the presence of the directly adjacent small segment. Thus numerous other suboptimal shorter segments were reconnected instead (Figure 5.18a and b). However, such a longer continuous segment possibly would have been identified if barriers were modified simultaneously in larger intervals (two or more at a time).

Even though the sequential procedure analysed in this thesis is not directly comparable to score-and-rank methodologies, the outcomes of both procedures highlight a sub-optimal trend in restoration. This suggests that restoring connectivity by selecting individual barriers one at a time detracts from locating the best possible solution for reconnecting the stream network. Understanding the implications of the number of barriers modified in a single restoration interval could provide valuable insight into which interval size yields near optimal connectivity restoration outcomes.

The procedure employed to select barriers within the Williams River to improve stream network connectivity could have significant implications to management. Sequential decision making by management in project implementation scenarios is of a high likelihood. Funds might become available sporadically or certain projects be implemented in multiple phases. Thus projects could be planned in tandem with fund availability by selecting the best current solution rather than being planned with a broader understanding and ‘master plan’. This thesis confirms the importance of advanced planning for restoration decision making and the assessment of potential future scenarios. For example, the type of procedure which is implemented has no significance if only very few modifications are to be carried out (Figure 5.16; Figure 5.17). On the other hand, a lack of advanced analysis and planning on larger or longer term projects can result in substantially sub-optimal outcomes for the available funds. On the Williams River up to 25 and 35 units of connectivity could be lost due to limited planning (Figure 5.16; Figure 5.18).

The results of this modelling approach suggests that implementing different barrier selection methods can result in substantially different levels of improvement, even when using an optimisation approach. Undoubtedly, the quality of outcomes and the effort placed into analysis are linked. Carrying out detailed analyses and investigating multiple scenarios in advance is more time consuming yet yields better results. However, this level of effort might be unnecessary. Thus it is important for management to clearly define project-specific goals to ensure a balance between maximum yield of funds and efficient use of time.

5.5 CHAPTER SUMMARY

This chapter has developed, tested and applied an optimisation model for the restoration of aquatic longitudinal connectivity in a network previously fragmented by the construction of multiple barriers.

Restoring connectivity of the Williams River network is sensitive and variable to both the stream network and barrier attributes. Multiple attributes influenced connectivity restoration, although each attribute influenced different components of the restoration process (level of connectivity and barrier selections), with some having more widespread and substantial impacts than others. Restoration of connectivity was the least influenced by the quality of patches making up the Williams River network. On the other hand, a change in barrier characteristics both substantially reduced connectivity and altered the composition of barriers selected for modification. This was further compounded by the variable cost of barrier modification. However, irrespective of the entrenched variability in connectivity restoration, network topology and the spatial configuration of barriers appeared to have an overarching influence on specific barrier selections, where ‘critical’ barriers were consistently targeted for modification. Critical barriers were predominantly located on the main channel of the Williams River network. Barriers located close to the river mouth were important for diadromous species behaviour, while centrally located barriers were important for potamodromous species behaviour.

Restoring connectivity of the Williams River network is dependent upon, and varies according to, species behaviour. Modifying a specific number of optimal barriers generally results in greater improvements in network connectivity when restoring for diadromous than potamodromous species behaviour. Furthermore, optimally selected barriers also differ between the two species behaviours. There is substantial difference when only a few barriers are modified, as the barriers most critical to restoring connectivity for each type of species behaviour differ. Restoring connectivity for diadromous species behaviour is more sensitive to barrier selections than potamodromous species behaviour. This is particularly evident when a small number of barriers are modified, where modifying a sub-optimal set of barriers can result in near insignificant improvements of stream network connectivity. Sub-optimal barrier selections also result in sub-optimal improvements in potamodromous connectivity, although this difference in connectivity is more prevalent when a larger number of barriers are modified. Thus, it is important to assess and consider both species behaviours when restoring stream network connectivity.

The type of barrier selection procedure implemented to restore stream network connectivity can have a substantial influence on the level of restored connectivity that is achieved. Selecting the optimal barriers to be modified in a sequence rather than wholly independently generally resulted in considerably suboptimal improvements in connectivity, except for early in the process when only a few barriers were modified. Thus, decisions that were previously made can exert a significant influence on future decision making, and detrimentally impact the potential level of improvement in connectivity that could be achieved. Understanding these interactions and the potential future implications of current decisions is of especially high significance to management authorities, who readily implement projects in multiple phases.

In the following chapter, the optimisation model developed for restoring stream network connectivity in Chapter 5 is applied to the entire Hunter River catchment (obtained from Chapter 4). It is implemented to identify the barriers in the catchment which are critical for maximising and restoring stream network connectivity for both species behaviours.

Chapter 6

6 Chapter 6 – Restoring Longitudinal Connectivity of the Hunter River Network

6.1 INTRODUCTION

Riverine landscapes have been extensively modified to secure and control variable water resources required for human needs (Arthington and Pusey, 2003). Barriers have been constructed on Australian waterways for this purpose (Williams and Watford, 1997; Arthington and Pusey, 2003). Change and reduction in connectivity is one of multiple ways in which river ecosystems have been altered (Tockner *et al.*, 2010). This has resulted in the loss and alteration of aquatic habitats (Harris, 1984) and subsequent reductions and alterations of natural ecosystem community structure (Gehrke *et al.*, 2002; Brainwood *et al.*, 2008). Initiatives aimed at conserving threatened ecosystems are being implemented as a result. These include specialised research on fish-friendly barrier design, barrier modification and monitoring of restored sites (Mallen-Cooper, 1992; Mallen-Cooper, 1994; NSW Industry and Investment, 2009; Boys *et al.*, 2012). Identifying regions considered critical for restoring connectivity within river networks is a significant step in advancing our understanding of where connectivity is to be maintained (Erős *et al.*, 2011). It provides a means of allocating limited management resources to regions for which the outcome may be substantial and where investment is maximised (O'Hanley and Tomberlin, 2005).

This chapter investigates restoring longitudinal connectivity of the artificially altered and fragmented Hunter River stream network. The most 'critical' anthropogenic barriers and regions for restoring and maintaining longitudinal connectivity in the entire Hunter catchment are identified. In addition, both the types of 'critical' barriers and the extent of improvement in connectivity that can be achieved through a number of optimal barrier modifications are explored. These analyses are carried out with the connectivity restoration optimisation model developed in Chapter 5. The strengths and weaknesses of the model as a tool for restoring connectivity are also assessed and described in relation to river science, other models and its useability and implications to management.

6.2 METHODS

The methods applied to identify the optimal number and locations of barriers for restoring stream network connectivity within the Hunter River are presented in this section. The data sources used to optimally restore longitudinal connectivity of the Hunter catchment, and the various stream network assumptions made in the optimisation model are presented. This is followed by a section which describes the optimisation procedure implemented and the techniques used for data analysis, in terms of assessing importance of all barriers and barrier types to stream network connectivity and respective change in connectivity character. Modelling and optimisation work was performed with *MATLAB R2011b* software package, while all spatial work was carried out in *ArcGIS Ver. 9.3*.

6.2.1 Data Source and Stream Network Model Assumptions

The data sources used as inputs to the optimisation model were obtained from previously developed ArcGIS shape files (Chapter 4). The first dataset utilised was a shape file of the stream network for the Hunter catchment fragmented by natural (Figure 4.4) and anthropogenic barriers (Figure 4.8), while the second was a merged shape file of the respective natural and anthropogenic barrier datasets. These datasets contained information on the spatial organisation of the stream network, in addition to the physical segment characteristics of the Hunter River network.

Development of an optimisation model for the Hunter River network involved a number of assumptions on the character of its stream network and the barriers within the network. Results from the sensitivity analysis carried out on the Williams River in Chapter 5 identified a ‘critical’ set of barriers which were consistently selected irrespective of the character of the stream network, its barriers and the number of modified barriers. The number of ‘critical’ barriers was equivalent to five percent of all barriers in the network. Accordingly, in identifying the ‘critical’ five percent of barriers of the Hunter River network, River Types of the network were not habitat quality weighted and all anthropogenic barriers were assigned the same characteristics. This is because of the limited influence these characteristics have had in determining the barriers selected to be modified earliest in the connectivity restoration optimisation procedure in Chapter 5. This refers to the group of barriers selected when only a low number of barriers are modified in comparison to the number of barriers found in the entire network. No information on barrier characteristics was available, as such each anthropogenic barrier was assumed to initially be fully impassable, while restoration at a unit cost resulted in a fully passable barrier with a porosity of one. These anthropogenic barrier assumptions are consistent with those used by Kuby *et al.* (2005). The network contains barriers which are natural in-stream obstacles with unknown passability. They were assigned a porosity of zero in accordance with the study of Cote *et al.* (2009). In restoring connectivity of the Hunter River network, natural barriers were considered natural features of the network and only anthropogenic barriers were modifiable.

6.2.2 Optimisation Procedure

Optimally restoring longitudinal connectivity of the Hunter River stream network was carried out using the generic optimisation model developed in Chapter 5. Restorations were carried out in relation to both diadromous and potamodromous species behaviour. Stream network data from *ArcGIS Ver 9.3* was reconstructed into the required format for the optimisation model with an automated tool in *MATLAB R2011b*.

Improvements in longitudinal connectivity of the Hunter River as a result of barrier modification were assessed for a sequence of restoration intervals so as to identify the important barriers and regions within the network for connectivity. Connectivity restoration was limited to a maximum of 125 barriers for both types of species behaviour, which is equivalent to five percent of barriers in the stream network. This maximum was done as a result of the model development undertaken in Chapter 5. The results from Chapter 5 suggest that the first five percent of all barriers modified were consistently selected irrespective of model assumptions and were responsible for considerably improving connectivity. At the coarser scale of restoration, barriers were modified in intervals of 25 at a time (Table 6.1). However, the first 25 barriers were also modified at a finer interval of five barriers at a time (Table 6.1). Commonly socioeconomic restrictions also only allow for limited restorations (Lake, 2001), hence restoring at a finer scale provides understanding of where restorations will most likely occur. In addition, the research of Kuby *et al.* (2005), O'Hanley and Tomberlin (2005) and O'Hanley (2011), all suggests that the most substantial improvements in connectivity occur following the removal of barriers selected earliest in the optimisation procedure. The independent optimisation procedure was then implemented to ensure that optimal decisions were made for each restoration interval, irrespective of any optimal decisions made at lower restoration intervals. All restored stream network connectivity values were scaled by the natural connectivity of the stream network, which is indicative of the maximum attainable level and then plotted as a function of the restoration interval.

Table 6.1 Interval characteristics used in the optimisation model for restoring longitudinal connectivity of the Hunter River network.

	Restoration Scale	
	Fine	**Coarse**
Restoration Budget Interval	5	25
Budget Range	5 to 25	25 to 125
No. of Restoration Intervals	5	5
Total Network Connectivity Restoration Cost	2466	

6.2.3 Data Analysis

6.2.3.1 Classifying Barrier Importance

The level of importance of each barrier for restoring longitudinal connectivity in the Hunter River was classified for both species behaviours at both the fine and coarse scales of restoration independently (refer to Table 6.1). The importance of a barrier for restoring connectivity was measured at each scale according to the proportion of intervals it was selected for modification, where each barrier could have been selected up to a maximum of five times and thus six classes. Accordingly, each barrier was placed into a class representative of barrier importance. Barriers which were selected to be modified at each interval were of the highest importance and classified under class 1. Barriers selected on four out of five occasions were in the next highest class of 0.8, barriers selected on three occasions were class 0.6, and so on. If a barrier was never selected it was classified under class 0, the lowest class of barrier importance. Barrier class data was imported back into *ArcGIS Ver 9.3* and the spatial layout of barrier importance and reconnected stream network was graphically represented in terms of barrier class.

Barrier composition between the two species behaviours was also compared for each restoration scale. This identified differences in barrier importance. Both the diadromous and potamodromous class of each barrier was identified, and accordingly the number of barriers which fell under each class ‘situation’ was counted.

6.2.3.2 Barrier Type Analysis

The importance of each of the four barrier types (dams, floodgates, weirs and road crossings) for restoring stream network connectivity was assessed for both species behaviours at both the fine and coarse scales of restoration. Barrier type datasets compiled in Chapter 4 were utilised to identify the type of each selected barrier of each class. After which, the total number of each barrier type of each class was calculated. This was followed by a calculation of the proportion of the total number of barriers selected of each respective barrier type.

6.2.3.3 Reconnected Segment Length Analysis

Data on the lengths of stream segments reconnected because of barrier modification were used to analyse the progressive change in connectivity for both species behaviour of the Hunter River network. Numerous representative variables of stream segment connectivity restoration were calculated for each interval. These included the total length of reconnected network and the average length of reconnected network per modified barrier as a result of the unique barrier selections. Additionally, the average length of reconnected network per modified barrier for the entire stream network was calculated and indicative of the above average outcome to modifying barriers by optimally targeting them.

6.3 RESULTS

The results of the optimal modelling of various barriers and regions in the stream network of the Hunter River are presented below in two main sections. Initially, the barriers important for restoring the Hunter stream network connectivity are spatially represented and their composition is also described. The second part of the section describes the change in stream network connectivity brought about by modifying the ‘critical’ barriers, and the variation in connectivity change associated with the number of barriers modified.

6.3.1 Critical Barriers of the Hunter Catchment

6.3.1.1 Critical Barriers, Level of Importance and Spatial Configuration

Barriers selected to be modified by the optimal modelling, were not evenly distributed across the five barrier classes (Figure 6.1). Barrier selections were variable for diadromous species behaviour where a total of 31 barriers were selected to be modified at least once at the fine restoration scale (refer to Appendix C for list of barriers). Furthermore, barrier classification between the classes was variable; barrier class 1 and 0.4 had the fewest barriers with a total of two, while 14 barriers were classified for barrier class 0.2. By comparison, barrier selections for potamodromous species behaviour were less variable. A total of 26 barriers were selected to be modified, with five barriers for classes 1 and 0.8, four barriers for class 0.6 and seven barriers a piece for classes 0.4 and 0.2.

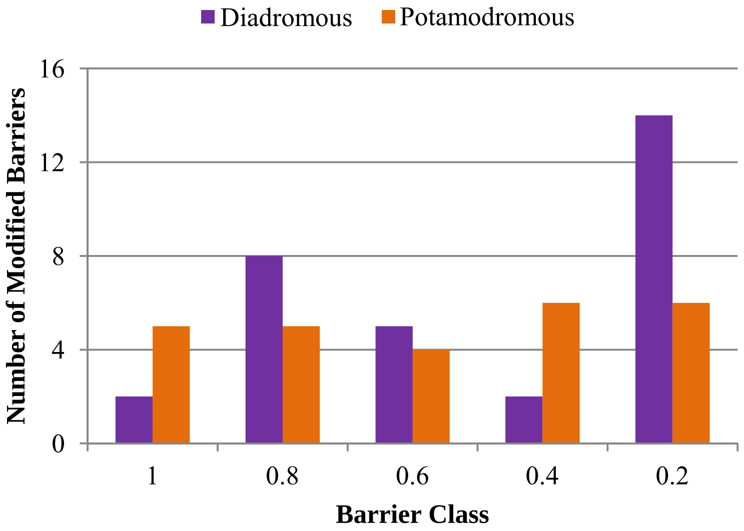
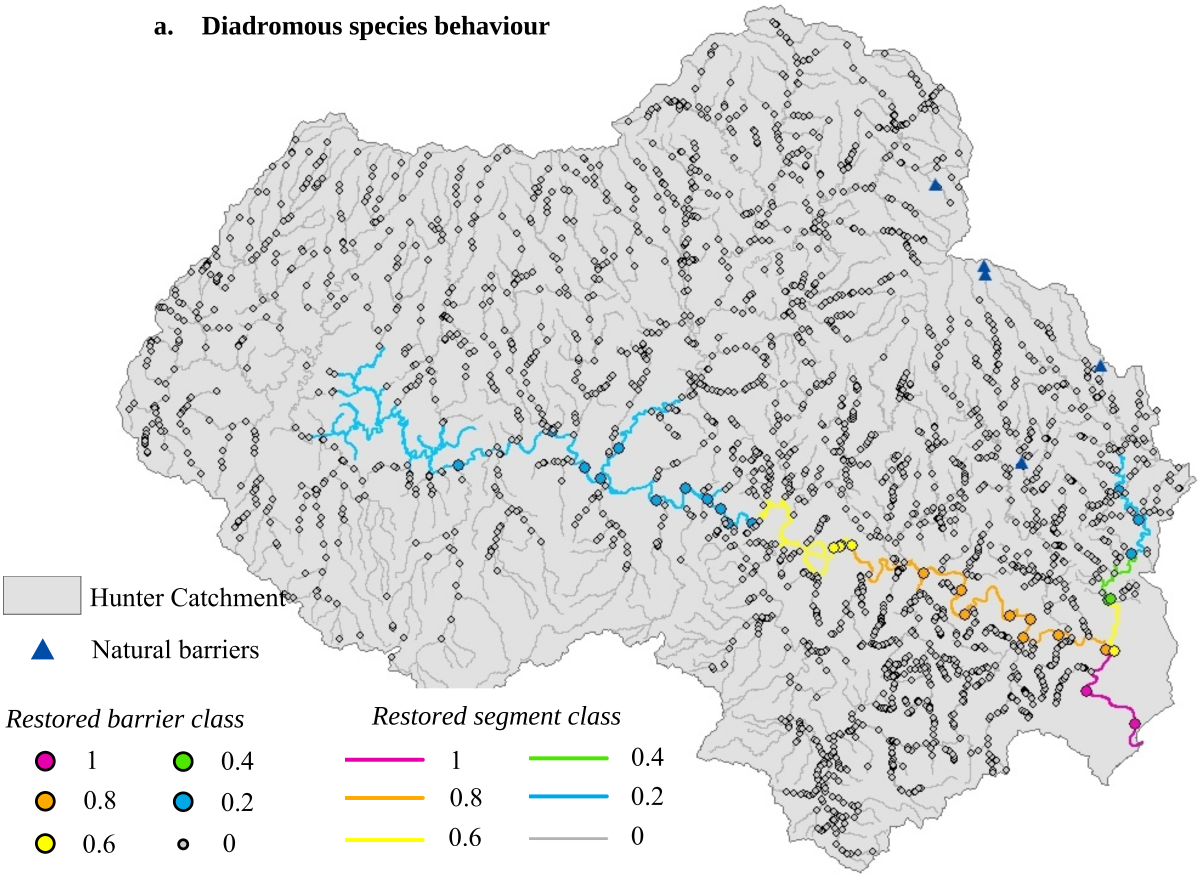


Figure 6.1 Number of barriers of each class, for both diadromous and potamodromous species behaviour at the fine scale of restoration.

At the finest scale of restoration, barriers critical to the optimal improvement of stream network connectivity exhibited broad patterns in terms of their spatial configuration (Figure 6.2). A general ‘radiating’ pattern was evident for both types of species behaviour, where barriers at the highest level of importance conglomerated and stream segments were

a. Diadromous species behaviour



b. Potamodromous species behaviour

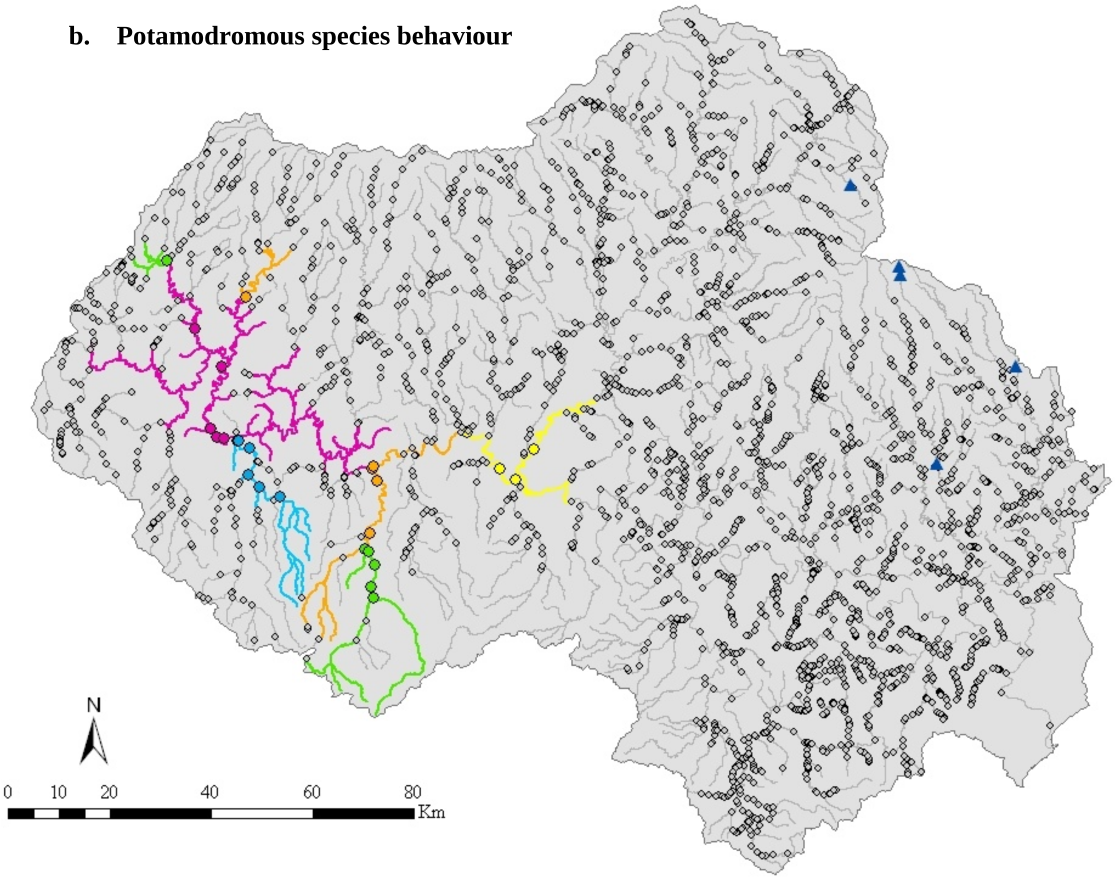


Figure 6.2 Spatial location of barriers of each class and the respective reconnected stream segments for a). Diadromous; and, b). Potamodromous species behaviour, at the fine scale of restoration.

reconnected in a radiating pattern away from it, with the order of importance of barriers (barrier class) descending in an outward direction (Figure 6.2a and b). The two barriers of class 1 for diadromous species behaviour were located furthest downstream in the region of the Hunter River estuary. Subsequent barrier selections expanded upstream, initially moving up the Hunter River, followed by the Williams River and lastly into the lower reaches of the Upper Hunter and Goulburn Rivers (Figure 6.2a). The most critical barriers for connectivity of potamodromous species behaviour were located on the Goulburn River (Figure 6.2b). Restorations followed into Widden Brook and Krui River, two tributaries of the Goulburn River. This was followed by further restorations downstream in the Goulburn and expanding into the Upper Hunter River, before barriers on Bylong and Munmurra River were selected.

Critical barrier selections for the two types of species behaviour were generally distinct and differed at the fine scale of restoration (Table 6.2). Only five barriers were selected for both species behaviours, where the barriers were of high restoration classes for potamodromous species behaviour but only class 0.2 for diadromous species behaviour (Table 6.2 – yellow cells). These barriers were located on the Upper Hunter River and on the downstream reaches of the Goulburn River (Figure 6.2a and b).

Table 6.2 Barrier composition for the diadromous and potamodromous species behaviours. The number of barriers falling under each of the six barrier classes for both species behaviours and the total number is indicated for the fine scale of restoration.

		Potamodromous Barrier Class						
		0	**0.2**	**0.4**	**0.6**	**0.8**	**1**	
Diadromous Barrier Class	**0**	2414	6	6	0	4	5	***2435***
	0.2	9	0	0	4	1	0	***14***
	0.4	2	0	0	0	0	0	***2***
	0.6	5	0	0	0	0	0	***5***
	0.8	8	0	0	0	0	0	***8***
	1	2	0	0	0	0	0	***2***
		2440	***6***	***6***	***4***	***5***	***5***	

At the coarser scale of restoration, where up to five percent of barriers were modified, there was lower variability in the barriers selected to be modified for the two species behaviours. A total of 126 barriers were selected at least once for optimal restoration (Figure 6.3; refer to Appendix D for list of barriers). The same trend was present for both migration types, where each of classes 1, 0.8, 0.6, 0.4 and 0.2 were composed of 25, 25, 24, 26 and 26 barriers respectively. Similar spatial patterns were evident for the selected barriers irrespective of species behaviour (Figure 6.4). Class 1 barriers occurred on the Hunter, Upper Hunter and Goulburn Rivers when connectivity was restored for diadromous species behaviour (Figure 6.4a). This was followed by further expansions of barrier class 0.8 upstream through the Goulburn River and into a few of its tributaries, Widden Brook, Bylong, Krui and Munmurra Rivers. Converse spatial patterns were present for potamodromous species behaviour restorations, where class 1 barriers were located on the Goulburn River, its tributaries and the Upper Hunter River, while barrier class 0.8 spread downstream through the Hunter into the Williams River (Figure 6.4b). Numerous class 0.6 barriers occurred in Wollombi Brook, after which restorations spread up the Upper Hunter River and into Doyles and Dart Rivers (class 0.4), while the least selected barrier class 0.2 were predominantly on Merriwa River and Parsons Creek (Figure 6.4a and b).

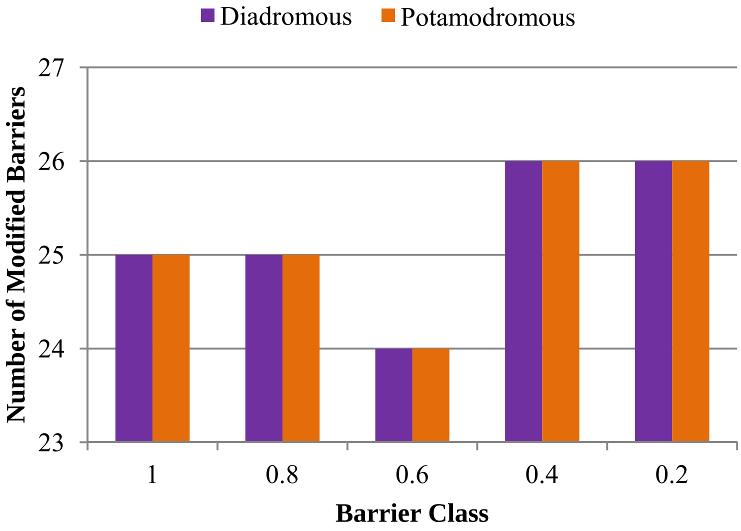
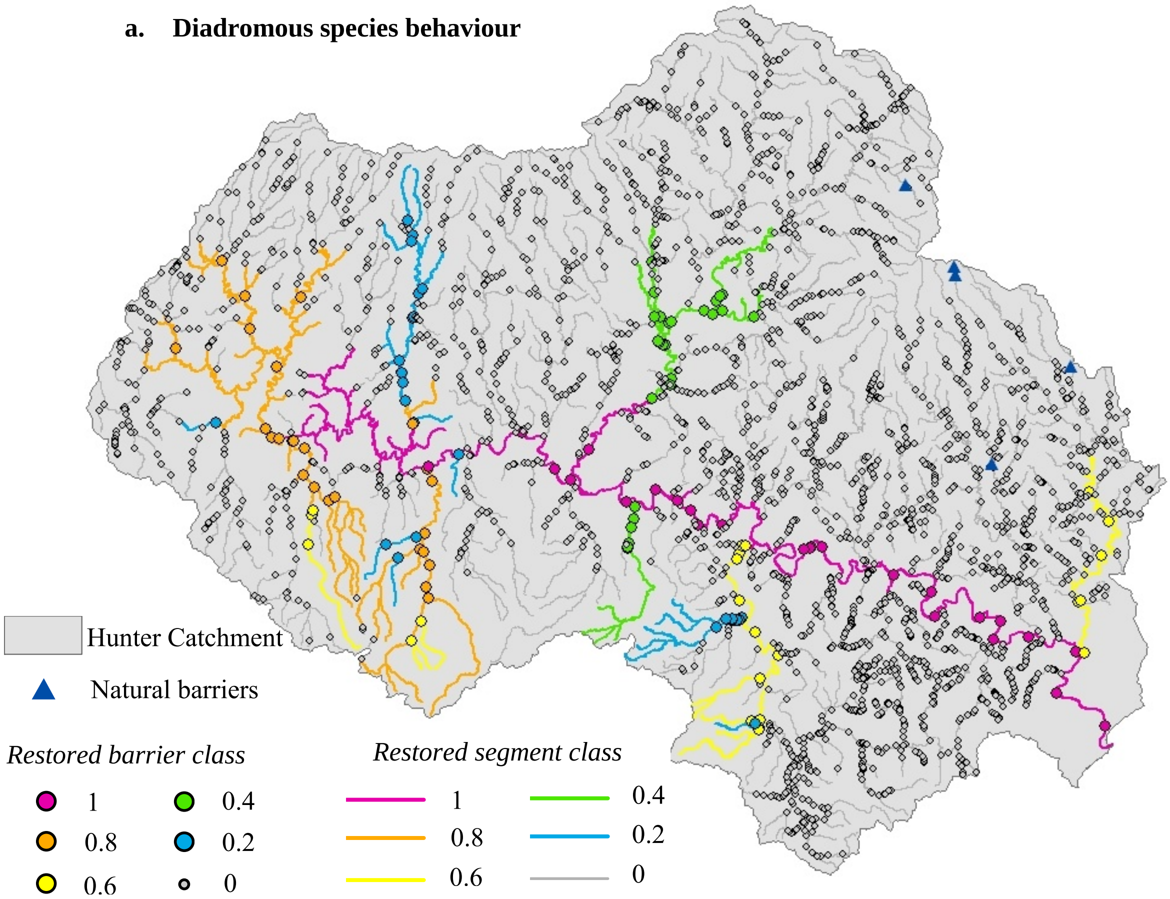


Figure 6.3 Number of barriers of each class, for both diadromous and potamodromous species behaviour at the coarse scale of restoration.

Critical barriers at the coarser scale of restoration were similar between the two types of species behaviour (Table 6.3). Only one of the 126 selected barriers completely differed between the migration types (Table 6.3 – orange cells). The barrier which was not selected to improve potamodromous connectivity was located furthest downstream in the entire network and critically ranked as class 1 for diadromous migrations (Figure 6.4a and b). Otherwise, there was a minor overlap between the more consistently selected barriers for the two species behaviours, where only five and three barriers were ranked under classes 1 and 0.8 irrespective of species behaviour. In general, the top priority barriers (class 1) for

a. Diadromous species behaviour



b. Potamodromous species behaviour

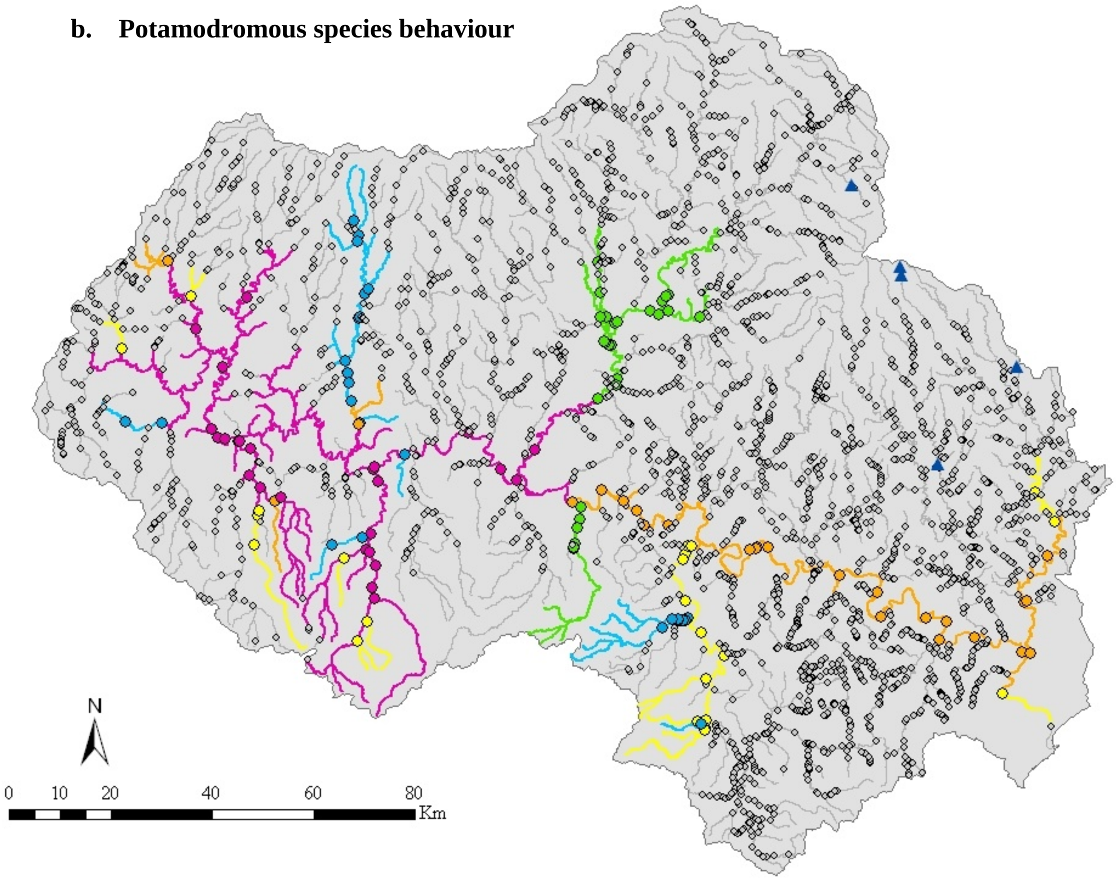


Figure 6.4 Spatial location of barriers of each class and the respective reconnected stream segments for a). Diadromous; and, b). Potamodromous species behaviour, at the coarse scale of restoration.

each species behaviour were only classified under class 0.8 for the opposite species behaviour (Table 6.3 – yellow cells). Differentiation between barrier compositions of the two species behaviours was significantly less for barriers of lower classes. A total of 20, 26 and 25 (out of 24, 26 and 26) barriers were the same for both diadromous and potamodromous migrations for barrier classes 0.6, 0.4 and 0.2 respectively.

Table 6.3 Barrier composition for the diadromous and potamodromous species behaviours. The number of barriers falling under each of the six barrier classes for both species behaviours and the total number is indicated for the coarse scale of restoration.

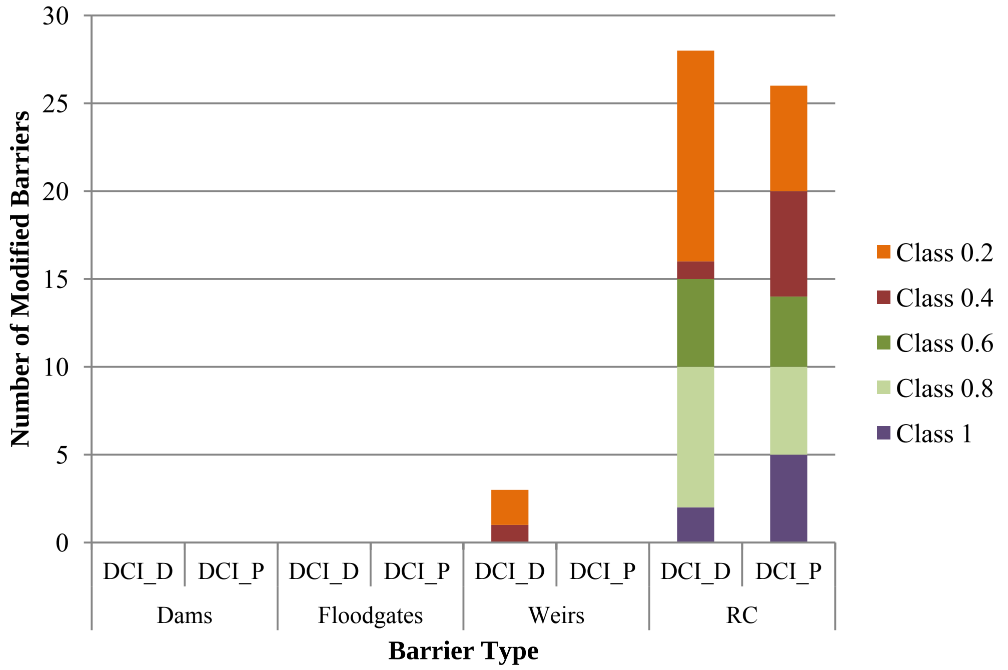
		Potamodromous Barrier Class						
		0	**0.2**	**0.4**	**0.6**	**0.8**	**1**	
Diadromous Barrier Class	**0**	2339	1	0	0	0	0	***2340***
	0.2	0	25	0	1	0	0	***26***
	0.4	0	0	26	0	0	0	***26***
	0.6	0	0	0	20	4	0	***24***
	0.8	0	0	0	2	3	20	***25***
	1	1	0	0	1	18	5	***25***
		2340	***26***	***26***	***24***	***25***	***25***	

6.3.1.2 *Critical Barriers and Barrier Types*

The types of barriers selected as being critical to stream network connectivity at the fine scale of restoration were predominantly road crossings in the Hunter catchment (Figure 6.5a, Appendix C). All 26 barriers selected for potamodromous species behaviour were road crossings, while for diadromous migrations 28 out of the 31 barriers were road crossings. Additionally, all barriers falling under classes 0.6, 0.8 and 1 were made up of road crossings. The other three barriers were weirs. Even though substantially more road crossings were selected as critical barriers than weirs, proportionally weirs are the most influential barrier type to stream network connectivity, with 0.3 percent in comparison to 0.012 percent of all structures being selected (Figure 6.5b). Seaham Weir on the Williams River was the highest priority of the selected weirs, with the other two structures being the gauging station on the Hunter River at Liddell and Jerrys Plain Weir.

Similarly, barrier selections at the coarser scale of restoration were also dominated by road crossing barrier types (Figure 6.6a, Appendix D). Road crossings made up 121 out of a total of 126 barriers for both diadromous and potamodromous species behaviour; in addition they also made up all the highest priority structures (class 1) for potamodromous species behaviour.

a. Total number of modified barriers



b. Proportion of modified barriers of each barrier type

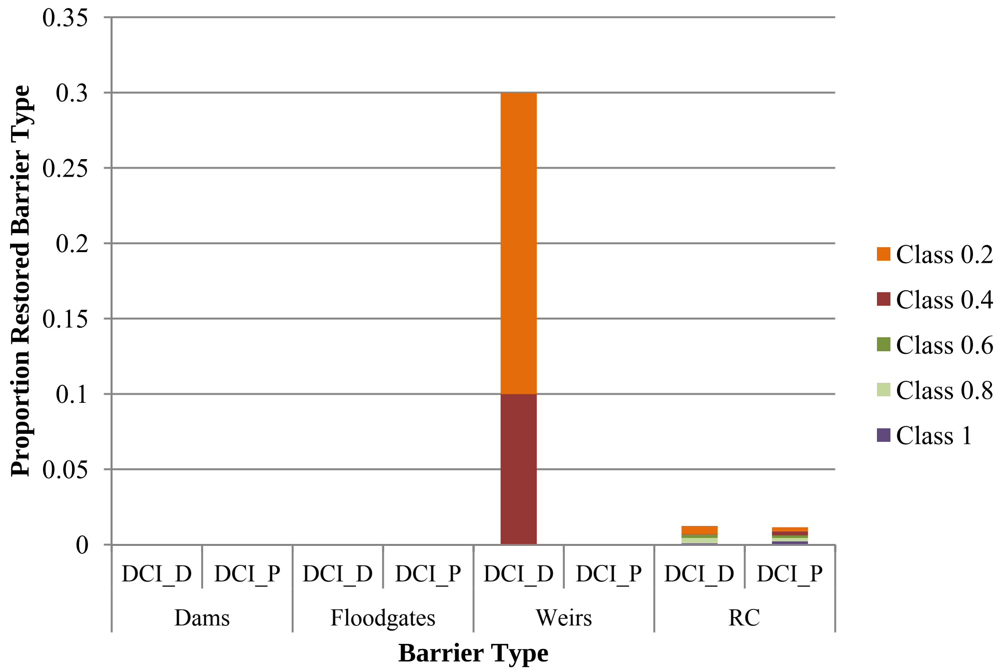
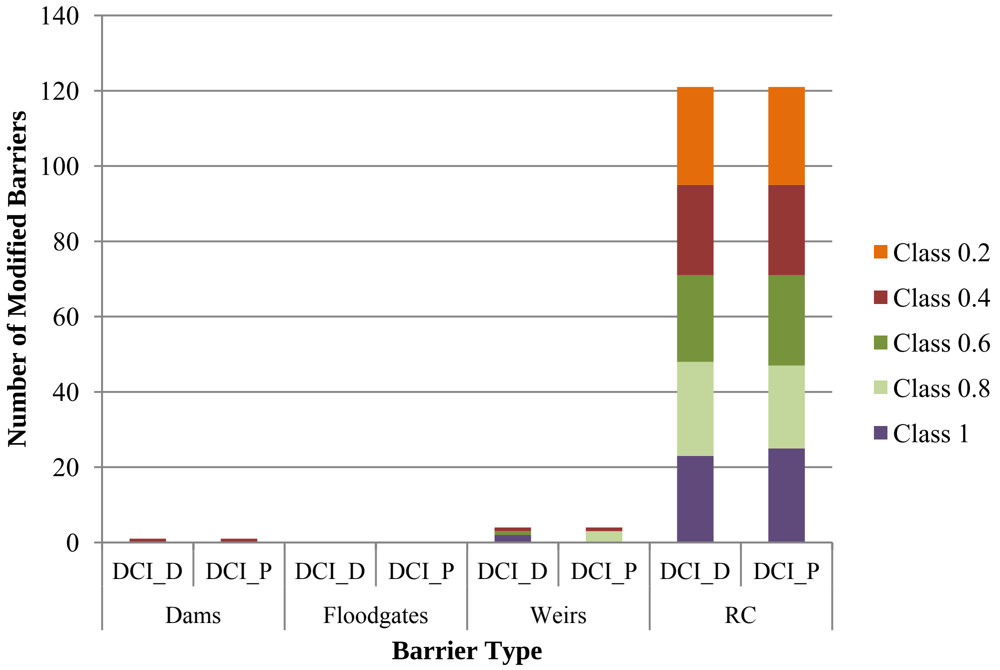


Figure 6.5 Distribution of a). The total number selected of each barrier type; and, b). Proportion of barriers modified of each barrier type, for each of five barrier classes and both species behaviour at the fine scale of restoration.

a. Total number of modified barriers



b. Proportion of modified barriers of each barrier type

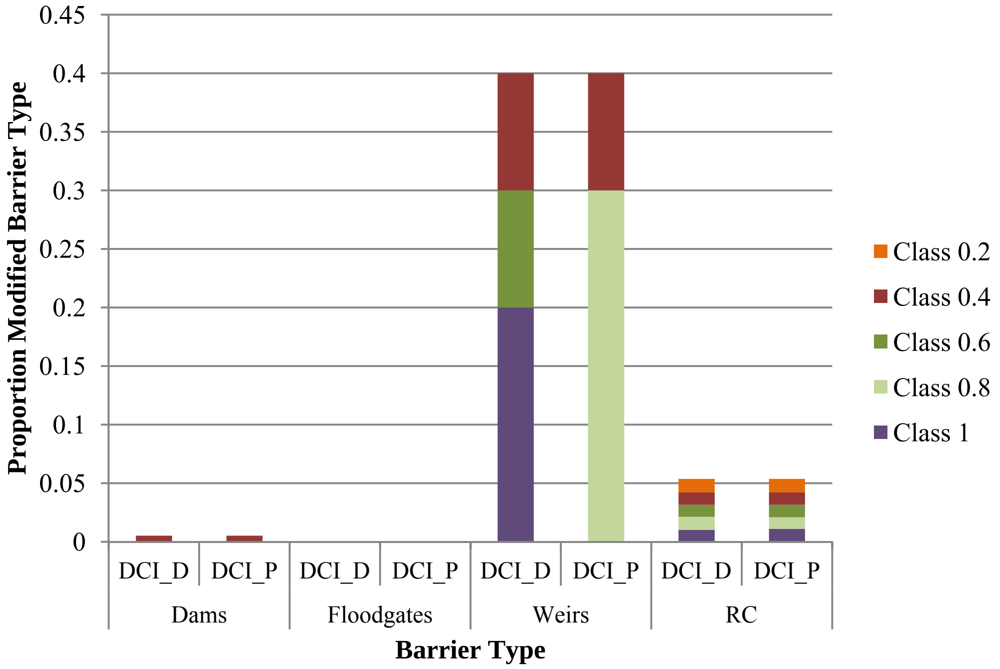


Figure 6.6 Distribution of a). The total number selected of each barrier type; and, b). Proportion of barriers modified of each barrier type, for each of five barrier classes and both species behaviour at the coarse scale of restoration.

No floodgates were deemed important in the early stages of restoring stream network connectivity, while only a single dam, Glenbawn Dam, was prioritised. The other four barriers were all weirs. The gauging station at Liddell and the Jerrys Plain Weir were of the highest priority (Figure 6.6a - diadromous: class 1, potamodromous: class 0.8), followed by Seaham Weir (D: class 0.6, P: class 0.8) and lastly the gauging station downstream of Glenbawn Dam (D and P: class 0.4). Even though road crossings were most abundant in total number, again, proportionally weirs were the most influential barrier type where 0.4 of the total structures were selected, followed by 0.054 of road crossings and lastly only 0.005 of all dams (Figure 6.6b).

6.3.2 Connectivity and Reconnected Stream Network

Modifying the optimal five percent of all barriers improved stream network connectivity for diadromous species behaviour, increasing from 0.078 to 23.027 (Figure 6.7). This reconnected a total of 2056.9 km of stream length at an average spacing of 16.455 km per modified barrier (Figure 6.8a). Restoring diadromous connectivity was marked by two distinct phases in the extent of improvement in connectivity (Figure 6.7). Improvement in connectivity occurred at a slower approximately linear rate for barriers selected earliest in the optimisation procedure, where at 20 barriers (0.81 percent) 322 km of stream was reconnected at a DCI of 3.605. The average length of reconnected stream varied between 15.06 and 16.21 km per barrier in this first phase. However, at one percent of modified barriers there was a substantial improvement in DCI to 6.37 and a jump in average stream length to 22.77 km per barrier. At this point restorations spread to the Goulburn River. This marked the start of the second phase in connectivity improvement, which was characterised by a logarithmic distribution where gradually the rate of improvement in connectivity decreased with increases in number of modified barriers. At two percent of modified barriers (50 structures), the average length of reconnected segment peaked to 24.45 km restoring a total of 1222.7 km of stream.

Restoration of connectivity for potamodromous life history exhibited a weak exponential trend, with DCI improving from 0.20 to 5.39 (Figure 6.7). Modifying five percent of barriers reconnected 2059.16 km of stream at an average of 16.47 km per barrier (Figure 6.8b). Converse to stream network connectivity, the total reconnected length of stream had a logarithmic distribution with improvement being highest at the start (Figure 6.8b). By modifying five barriers, 403.33 km of stream were reconnected at an average of 80.67 km per barrier. There was a rapid reduction in reconnected average stream length until one percent of barriers were modified, after which the rate of change stabilised and decreased. From approximately two percent of modified barriers, the improvement in connectivity is similar between both species behaviours, with maximum difference in average stream length of 0.2 km per barrier (Figure 6.8a and b).

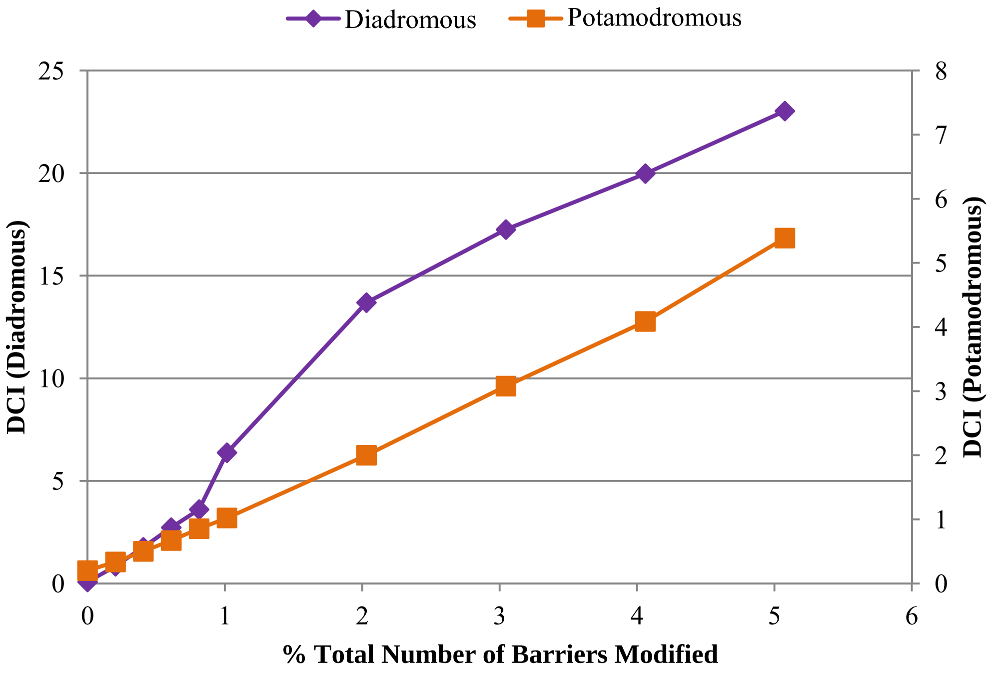
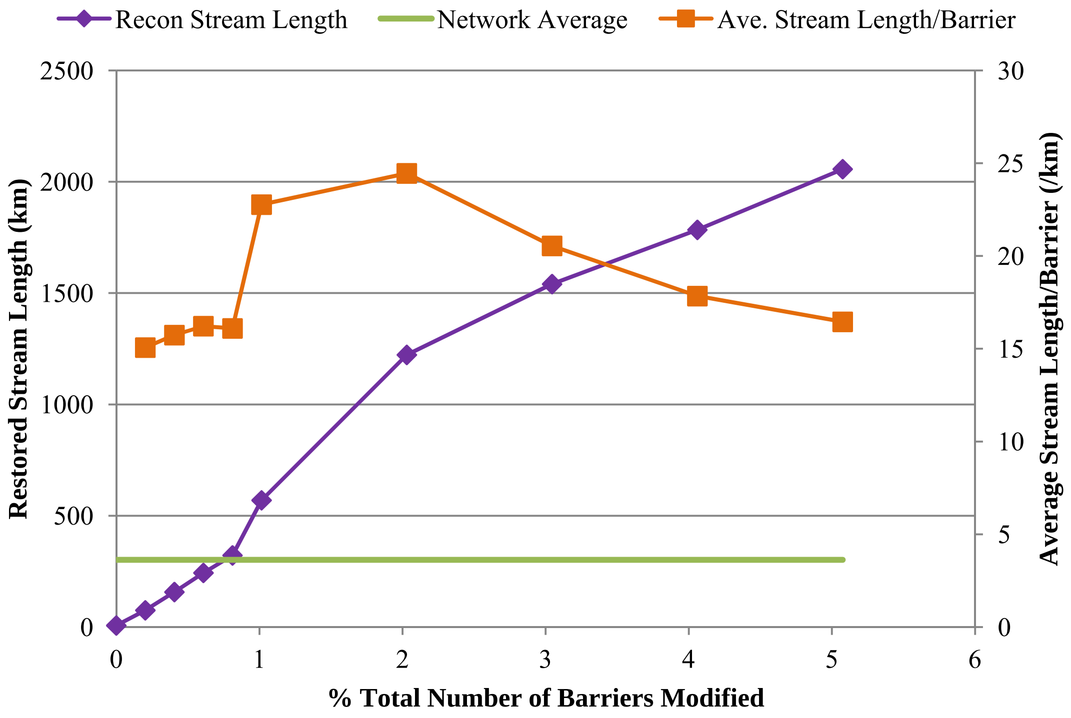


Figure 6.7 Graphical representation of the optimal improvement in stream network connectivity when modifying up to the top 5 percent of all barriers, for both diadromous and potamodromous life histories. The DCI values were scaled by the maximum achievable fully restored connectivity values (DCI_D = 98.33 and DCI_P = 96.72).

a. Diadromous species behaviour



b. Potamodromous species behaviour

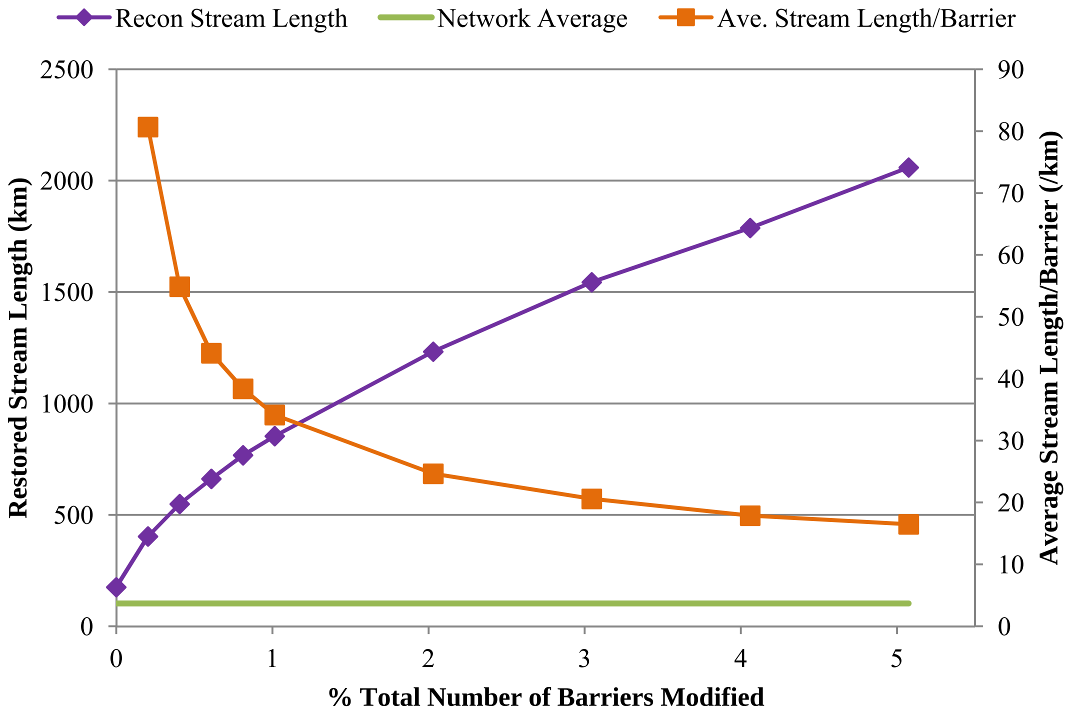


Figure 6.8 Distribution of reconnected stream segment data, namely total reconnected stream length, average length of reconnected stream per barrier and the total network average of restored stream length per barrier, as a function of the proportion of modified barriers, for a). Diadromous; and, b). Potamodromous species behaviours.

6.4 DISCUSSION

6.4.1 Barriers Critical to the Connectivity of the Hunter River Network

Restoration to achieve optimal connectivity in a highly modified network must be cognisant of the number and type of barriers as well as their location in the network in context of the reasons for restoration.

Locations of the barriers which are modelled as being critical to restore stream network connectivity of the Hunter catchment exhibited a broad spatial pattern. The main river channels were significant to the connectivity of the entire network. The two longest rivers, the Hunter and Goulburn, have the highest priority, with each being important to both diadromous and potamodromous species behaviours (Figure 6.2 – Class 1). This finding is similar to that reported by Cote *et al.* (2009), who identified downstream barriers as most critical to restoring diadromous connectivity, while barriers which were centrally located in the stream network were significant to potamodromous connectivity. In this study of the Hunter River, the major tributaries of the Williams River, Wollombi Brook and Upper Hunter River, also were important to network connectivity (Figure 6.4). Similarly, Eros *et al.* (2011) found that high order streams were more important to maintain the connectivity of Zagyva River catchment, Hungary. Numerous smaller streams generally located in the south-western sections of the Hunter catchment were also significant to its connectivity (Figure 6.4). These streams are fragmented by relatively few anthropogenic barriers and have extensive stream systems, which could be linked to the lower levels of development in this region.

Many studies have highlighted the extent to which rivers have been disrupted and subsequently fragmented by anthropogenic barriers (e.g. Harris, 1984; Nilsson *et al.*, 2005; Walter and Merritts, 2008), as well as the effect these barriers have on local ecosystems (e.g. Ward and Stanford, 1983; Warren and Pardew, 1998). There have been fewer studies that have investigated the means and success of reducing their impacts, although MacDonald and Davies (2007) and David and Hamer (2012) are two out of a number of exceptions. However, studies on identifying and prioritising barriers important for improving network connectivity remain limited.

Various barrier types in different abundances were selected as critical to achieving substantial improvements to connectivity in the Hunter River network. Road crossings were the overwhelmingly dominant barrier type deemed critical to restoring connectivity (Figure 6.5; Figure 6.6). 'Bringing Back the Fish' project (NSW Industry and Investment, 2009) is the only other known study that identified and modified barriers of different types across the coastal catchments of NSW, with the aim of improving fish passage and connectivity. The restoration efforts of this study were predominantly focused on floodgates located in the lower catchment, contrasting the findings of this thesis. These differences in barrier selections between the two studies can potentially be attributed to a number of reasons: difference in utilised prioritisation techniques, differences in scales of restoration (single as to multiple catchments) and numerous other external unaccounted for factors that were experienced by NSW Industry and Investment when implementing the restoration projects (NSW Industry and Investment, 2009).

The dominance of road crossings as the most important barrier type for fish movement has important implications for the design, management and general understanding concerning the restoration of stream network connectivity. It is well established that dams have brought about major changes to riverine landscapes and the ecosystems they support (Bunn and Arthington, 2002). Consequently much debate has arisen around dam removal (Stanley and Doyle, 2003). However, the results in this thesis highlight the potential importance of road crossings and run-of-river water retaining structures (Liddell Gauging Station and Jerrys Plain Weir), which were consistently and more regularly selected than larger barriers. These findings suggest that smaller barriers may be more critical in restoring stream network connectivity than larger dams. Individually, the effects of small barriers are minute in comparison to large structures, which commonly act as permanent barriers to many species. However, the cumulative effects of small barriers may be more significant (Thoms and Walker, 1993), although the influence of larger structures should not be ignored. Furthermore, these findings emphasise the importance of designing road crossings according to guidelines with explicit provisions for fish passage (e.g. Cotterell, 1998; Fairfull and Witheridge, 2003; Kapitzke, 2010).

Restoring connectivity in the Hunter River and other coastal basins of New South Wales is a management priority for the region (HCRCMA, 2013a). Fish passage can be improved by either removing in-stream barriers, altering them to improve porosity, or through better management of flows to enhance the porosity of barriers (NSW Industry and Investment, 2009). Hydrological connectivity and tidal flushing was increased in the estuarine wetland regions of Hexham Swamp through the opening of floodgates (HCRCMA, 2013b). There is already evidence of positive ecological responses to increased connectivity (e.g. Boys *et al.*, 2012). The results of this thesis provides valuable information to regional catchment management authorities on where in the catchment stream network connectivity restoration efforts, as a result of barrier modification, should be focused (Figure 6.2; Figure 6.4), while also informing on potentially valuable ‘break-points’ in connectivity improvement that can be achieved (Figure 6.7; Figure 6.8a). Furthermore, these findings could be coupled with other conservation efforts that have and currently are being implemented in the catchment, including riparian revegetation (Harris *et al.*, 2012), resnagging (Brooks *et al.*, 2004), wetland reconnection (NSW Industry and Investment, 2009) and environmental flow releases (Rolls *et al.*, 2011). The multitude of restoration projects implemented across the catchment provides an ideal opportunity to improve various components of the riverine landscape and increase ecological response by developing a network of interrelated mini-projects, such as in the Snowy River, Victoria (Lake, 2005). Restoring one to a number of parts of the riverine landscape in a certain region could also prove useful in testing ecological theories and the influence of various and multiple drivers on ecosystem structure and function.

6.4.2 Dendritic Connectivity Index – A Tool for Optimising Stream Connectivity Restoration

A new approach to optimally restore longitudinal stream network connectivity was developed and applied in this thesis. The Dendritic Connectivity Index (Cote *et al.*, 2009), which is based on landscape ecology concepts of connectivity and provides a means to quantify it, was utilised as a measure of stream network connectivity. Connectivity within landscapes was described by Wiens (2002) as a critical landscape attribute, which was a function of a complex set of interactions between patch quality, boundary properties, patch context, movement characteristics of the feature of interest and distance between locations.

The habitat patch quality adjusted DCI utilised in this thesis incorporated a multitude of riverine landscape and species attributes to assess connectivity. These included patch quality and configuration, all natural and anthropogenic longitudinal barriers, species type, species life history and mobility characteristics at different life stages. The metric was applied to the Hunter catchment and Williams River sub-catchment. It is equally applicable to river systems of various spatial scales, from large catchments to sub-networks of small river systems. The flexibility of the index makes the optimal connectivity restoration approach as applied in this thesis, robust and pertinent to a variety of fragmented river systems and species. For example, in the south-eastern coastal catchments of Australia, barrier design research has been specifically targeted at ensuring upstream passage of juvenile Australian bass (Mallen-Cooper, 1992). In the Murray-Darling Basin there have been attempts to facilitate upstream passage of adult golden and silver perch (Mallen-Cooper, 1994). The approach applied in this thesis is equally suitable to both scenarios. In addition, the model can be specifically altered to incorporate movement patterns and habitat requirements of the target species. For example, Australian bass have been found in streams up to 600m in altitude (Harris and Rowland, 1996). Thus, streams higher than 600 m could be given a zero habitat quality weighting in the model. Furthermore, the sensitivity of the model to a multitude of conceptual connectivity characteristics makes it more appropriate for restoring connectivity than most traditional approaches, which commonly implement highly simplified and subjective barrier prioritisation techniques (e.g. Poplar-Jeffers *et al.*, 2009). However, in applying the model, it is important and valuable for the user to understand what species connectivity is being restored and its behavioural characteristics, whether it is a single species or even a specific life-stage, to the entire fish community.

A number of optimisation models similar to that applied in this thesis have been developed to assist in river ecosystem management. The general method implemented in this model was to select the optimal combination of barriers for improvement (or removal) in relation to a desired outcome (connectivity in this case). This generally was the same technique employed by all the models identified in the literature (e.g. Kuby *et al.*, 2005; O'Hanley and Tomberlin, 2005; Zheng *et al.*, 2009; O'Hanley, 2011). In terms of structure, the model utilised in this thesis was similar to that by O'Hanley and Tomberlin (2005). However, the objective of their model was to increase habitat availability for anadromous species by the improvement of small barriers. Their model did not directly address restoring connectivity of an entire stream network, whether naturally or artificially fragmented. It was also only suitable for a single type of species behaviour, thus having limited use to south-eastern Australian riverine landscapes which support ecosystems of various species behaviours (McDowall, 1996). A more recent model by O'Hanley (2011) optimised barrier selections for reconnecting the longest individual segment in a stream network. As such, it is not prejudiced to any species behaviour unlike the previous model. However, this also means that species behaviour cannot be incorporated into the model if desired. Additionally, it the model does not incorporate any barrier porosity characteristics. The model developed in this thesis incorporated numerous critical stream and barrier characteristics (Wiens, 2002) and merged various aspects identified as important in other optimisation models.

Computation time is an issue of optimal models (O'Hanley and Tomberlin, 2005). As the number of barriers in a stream network increased so did the time required to calculate the DCI value of the system. This is a significant weakness of the model, especially in relation to potamodromous species behaviour, where the number of sub calculations carried out in quantifying the DCI value increases quadratically with an increase in the number of barriers (Equation 5.2). This can result in slow solving times for systems with many barriers. Furthermore, the numerical significance of each sub calculation (probability of movement between any two segments) to the final DCI value also substantially decreases for systems with more barriers. This is especially true when calculating movement between segments which are separated by a larger number of barriers. For example, if all barriers between two segments have 50 percent porosity, the probability that movement can occur between segments separated by at least seven barriers is less than one percent (Equation 5.6). Hence, significant lengths of computation time are spent on near insignificant values, unnecessarily slowing the time required in solving the problem. Approximation theory is a field of study concerned with replacing more complicated mathematical functions with simpler expressions (Cheney, 1966), and could potentially prove beneficial in reducing DCI computation times. Development of an approximate function of the DCI metric could be a useful future study. The issue of computation time is also further, and unnecessarily, exacerbated by barrier datasets with a significant number of fully porous structures that are included in the computations but have no effect on the connectivity value. Refining barrier datasets by removing potential barriers that have been identified as fully porous structures would reduce computation times.

The modelling approach adopted in this thesis is essentially a desktop exercise. Hence, it is suitable to those regions and river systems for which digital information on the drainage network and barriers is available, making it a simple and viable tool for river management across various regions and spatial scales. Traditional approaches that have been utilised in prioritising barriers for restoring connectivity have commonly involved intensive field data collection on barriers as an initial step in the prioritisation procedure (e.g. NSW DPI, 2006b; Poplar-Jeffers *et al.*, 2009; Kemp and O'Hanley, 2010; Nunn and Cowx, 2012). Even though such levels of detail are valuable for informed river management, carrying out widespread fieldwork can be both costly and time consuming. This is particularly true in larger regions such as the Hunter catchment where 2466 potential barriers were identified. In addition, it is more than likely that only a limited number of the examined barriers will ever be modified due to various socioeconomic restrictions. Furthermore, depending upon which field assessment technique is employed (refer to Kemp and O’Hanley (2010) for examples), additional desk-based analyses and modelling work might be required to process the data. These additional analyses might not necessarily even produce accurate and valuable outputs (e.g. Burford *et al.*, 2009). The modelling approach applied in this thesis can utilise this commonly collected barrier field data to prioritise optimal barriers for restoring connectivity. In addition, a further advantage of this approach is that the model can be used to explore what regions and barriers in the system are important for connectivity by running multiple hypothetical scenarios with minimal data input. Thus the amount of fieldwork and associated costs can be limited only to the most important barriers. The additional information collected in the field can be used to refine the output quality of the model, remove potential barriers which have been identified as fully porous and reduce computation times.

6.5 CHAPTER SUMMARY

The barriers and regions in the network critical to optimally restore longitudinal connectivity in the Hunter River network were identified. Barriers located on the main waterways were of the highest importance, with the Hunter River being of particular significance for diadromous species behaviour while the Goulburn River was central to restoring for potamodromous species behaviour. Barriers located on the other major waterways in the catchment, including the Williams River, Wollombi Brook and the Upper Hunter River, were also important to the connectivity of the stream network. Furthermore, the streams in the largely natural and under-developed south-western sections of the catchment also appeared to be vital. The majority of barriers identified as critical for connectivity restoration were of the road crossing barrier type, with this barrier type being consistently selected across both species behaviours and restoration scales. However, weirs also appeared to be of substantial importance to restoring connectivity, even though they were significantly less abundant.

Barriers critical to achieving optimal improvements in the connectivity of the Hunter catchment were identified with the use of a connectivity restoration optimisation model. The modelling approach that was adopted is largely desktop based. Thus, it is suitable to numerous regions and river systems worldwide for which digital information on the stream network and respective barriers was available. The model that was implemented to quantify stream network connectivity incorporated a number of landscape and species characteristics which have been recognised as important drivers of landscape connectivity. Thus, the model used in this thesis improves on other models which are limited in these spheres. This approach has substantial relevance to management, as it identifies which barriers are optimal for restoring connectivity at a catchment scale, but can also be utilised as a tool for exploring key regions and gaining understanding on important characteristics of connectivity. Future studies should investigate how to improve the required computation time for DCI calculation and problem solving. Furthermore, incorporating species dispersal abilities in quantifying connectivity would aid in improving our understanding of connectivity restoration of entire catchments.

Chapter 7

7 Chapter 7 – Synthesis

7.1 INTRODUCTION

Fragmentation of riverine landscapes has compromised the integrity of these ecosystems (Nilsson *et al.*, 2005). Improving the longitudinal connectivity of riverine landscapes through the removal of the barriers, physically altering them to improve their porosity and even changing their operational management has all become a common means of rectifying these stressors on river ecosystems. The need for restoration to be undertaken within larger catchment scales is increasing in prominence (Bohn and Kershner, 2002; Wohl *et al.*, 2005). However, knowledge and approaches to restoring connectivity of entire stream networks is limited. This thesis addresses the knowledge gap concerned with restoring longitudinal connections of an entire stream network. It does so with the use of optimisation methods, demonstrating they are an effective decision support tool for the analysis of complex systems. There are three main components to this thesis in which optimally restoring longitudinal stream network connectivity was addressed. First, the thesis examined the extent and nature of change in the longitudinal connectivity of the Hunter River network caused by the construction of in-stream barriers. Second, a model to target optimal restoration of connectivity was developed and its sensitivity in terms of influencing outputs to system character was also assessed. Third, this model was applied to identify those barriers and regions of the stream network important to restore connectivity within the Hunter River. The model utilises the Dendritic Connectivity Index, a new measure of longitudinal connectivity in stream networks (Cote *et al.*, 2009). Model outputs demonstrate the efficacy of the approach for identifying where restorations should occur within the stream network of the Hunter River, while simultaneously providing a means to investigate the influence of multiple drivers on network connectivity and thus potential restoration efforts. This thesis contributes to the science of river restoration by providing a new perspective on assessing and restoring longitudinal connectivity to river systems at a catchment scale rather than focusing at the traditional site scale.

This concluding chapter has four sections. First, the overall model outputs as they relate to restoring connectivity of the Hunter River are presented. Second, advances in understanding of the restoration of river systems at a catchment scale are discussed along with the benefit of using optimisation methods. Third, the significance of the findings and approach undertaken to the management of river systems is discussed. Finally, opportunities for future research are presented in light of the limitations of the approaches undertaken in the thesis are outlined.

7.2 RESTORING LONGITUDINAL CONNECTIVITY OF THE HUNTER RIVER NETWORK

Improving longitudinal connectivity of stream networks is intricately linked to the number and location of barriers selected to be physically modified (Cote *et al.*, 2009). Generally, the largest improvements in stream network connectivity occurred following the modification of barriers selected earliest in the optimisation procedure. The rate of change in this improvement then decreased as a larger number of barriers were modified, until modifying the final few barriers in the network only achieved minor gains in stream network connectivity. The findings of this thesis suggest that if a suboptimal set of barriers is selected, whether this is because of restoring for the incorrect species behaviour, utilising a sequential optimisation procedure or some other reason, then gains in stream network connectivity can be substantially lower, often to the extent of being insignificant at a catchment scale. This suggests that substantial gains in the longitudinal connectivity of a system can be achieved by modifying a limited number of barriers, if the barriers are specifically targeted in relation to the entire network. Otherwise, modification of inappropriately chosen barriers can be insignificant in terms of the connectivity of the entire network, even if the same number of barriers is modified.

Restoring longitudinal connections within stream networks through optimal barrier selections is variable and dependent upon the character of both the riverine landscape being restored and the barriers to be modified. Results of the optimal model for the Hunter River suggest that restoring connectivity is least influenced by the habitat quality of the different River Types that make up the stream network. Habitat quality evoked minor variations in achieved levels of restored connectivity and the selected barriers. By comparison, boundary characteristics, especially barrier porosity, appear to control the level of improvement of stream network connectivity, while simultaneously increasing differences between selected barriers. However, unequal barrier modification costs exacerbate variation in successive barrier selections. Despite the variability in connectivity restoration because of system attributes, the spatial organisation of the stream network and its barriers appear to be significant factors in determining optimal barrier selections. A number of 'critical' barriers were consistently selected to be modified in the Williams River, irrespective of the other system attributes. Diadromous species migrate between freshwater and saline environments as a part of their life-cycle; consequently barriers located in the downstream reaches of the Williams River sub-catchment were ecologically significant in restoring connectivity for diadromous species behaviour and maintaining the foremost migration routes. On the other hand, potamodromous species migrate wholly in freshwater, with improvement in connectivity being greatest when more centrally located barriers are removed. This lack of variation is significant as it appears to identify the most crucial barriers to optimally restore stream network connectivity, and thus the connectivity of the Hunter River.

The connectivity of the Hunter River has been substantially fragmented because of the presence of 2 466 anthropogenic barriers. These were present as weirs, dams, floodgates and road crossings. These barriers occur in different abundances and are unevenly distributed throughout the stream network, of which a combination of both characteristics influences the extent of change in connectivity that is exerted on the entire river system. Barriers located on the Hunter River and the Goulburn River (including a number of its tributaries) were identified as central and crucial to optimally restore connectivity in relation to both diadromous and potamodromous life histories. Furthermore, modifying barriers located on majority of the main waterways and in the sparsely-fragmented, underdeveloped western regions of the drainage basin also provides further significant improvements to the general state of connectivity of the basin, especially in relation to diadromous life history. Road crossings and weirs are the most critical barrier types for connectivity restoration, and also appeared to have the largest individual impact in reducing the basin's connectivity. This is likely a result of their location, with both barrier types being present on the main waterways and further downstream in the basin, unlike dams and floodgates. Modifying this limited number of optimally targeted barriers would result in substantial increases, well above the barrier average, in Hunter Basin stream network connectivity.

7.3 OPTIMALLY RESTORING LONGITUDINAL CONNECTIVITY OF STREAM NETWORKS

This thesis presents a shift in spatial scale at which the connectivity of riverine landscapes is examined. Traditionally, restoration of riverine landscapes has been concerned with smaller spatial scales (Hillman and Brierley, 2005); however, the focus of this thesis is on the entire river catchment. The means by which improvements to longitudinal connectivity at site or reach scales have been substantially investigated, commonly involving barrier removal (Doyle *et al.*, 2005), operational (Boys *et al.*, 2012) or structural alterations (David and Hamer, 2012) to the barrier. However, catchments are increasingly being considered to be the principal unit of investigation of hierarchically nested riverine landscapes (Chorley, 1967; Petts and Amoros, 1996; Thorp *et al.*, 2008), and the importance of carrying out restorations within a catchment context has been highlighted especially considering the top down constraints the hierarchical structure of catchment systems imposes (Wohl *et al.*, 2005; Harper *et al.*, 2008). Despite the increasing shift towards catchment scale approaches in restoration, knowledge at this scale remains limited. The findings of this thesis elucidate the importance and variability of different system attributes in restoring connectivity, which stream network topology and life-cycle migratory requirements of species appear to substantially dictate. This is especially evident for diadromous species which undergo compulsory migrations between fresh and saline water as a part of their life-cycle. In addition, the major rivers in a catchment are pivotal to restoring connectivity of the stream network. Furthermore, this thesis indicates that restoring stream network connectivity is closely associated to the composition of selected barriers. Thus, the commonly implemented technique of selecting restoration sites opportunistically (Alexander and Allan, 2007) is undesirable. While such restorations may reconnect a substantial amount of stream at a reach scale, they may result in negligible improvements in connectivity of the stream network. This thesis is significant as it provides new knowledge on patterns and system attributes which influence restoring longitudinal connectivity of entire stream networks. It also appears to be the only detailed study investigating restoration at a basin scale, rather than the more traditional site or reach scale.

Optimisation theory was applied in this thesis to optimally restore the longitudinal connectivity of a stream network. A modelling approach was utilised to identify the ideal combination of barriers to be modified so as to maximise improvements in stream network connectivity within project budgetary constraints. The mathematical formulation of the stream network connectivity restoration model is broad and robust, allowing for numerous system attributes to be incorporated in an analysis. For example, the model can account for different types of freshwater species, their habitat preferences and behavioural attributes at various life stages. Furthermore, different types of natural and anthropogenic barriers, such as waterfalls, weirs, floodgates, dams and road crossings, and their individual porosities can be incorporated. Additionally, the impact to connectivity of barriers that are not modifiable can even be considered. Optimisation is a beneficial numerical tool that allows for quantitative systems analysis to be carried out on complex systems. It has had widespread application in fields of engineering design and operations research, and has more recently been utilised to investigate both terrestrial and aquatic natural environments (e.g. Larson *et al.*, 2003; Stralberg *et al.*, 2009). This thesis has contributed to the application of optimisation theory by providing quantitative understanding and a platform for informed decision making on restoring connectivity in stream networks. The model used provides an alternative to the more traditional score-and-rank techniques applied in barrier prioritisation. In addition, the model has expanded on other riverine optimisation studies (e.g. Kuby *et al.*, 2005; O'Hanley and Tomberlin, 2005; Zheng *et al.*, 2009; O'Hanley, 2011) by increasing its applicability to various different systems. Moreover, this thesis also highlighted the value of less commonly employed optimisation solution algorithms, such as genetic algorithms, to solve complex riverine landscape problems.

7.4 SIGNIFICANCE OF THESIS TO MANAGEMENT OF RIVER SYSTEMS

The findings of this thesis have significance for management of the Hunter River catchment. The natural resources and services Australia's river systems provide are valuable and there is substantial monetary investment in their conservation and restoration

(Brooks and Lake, 2007). Reduced connectivity and habitat availability as a result of in-stream barriers has long been identified as a problem (Harris, 1984), and government agencies have recently invested into improvement of fish passage along the coastal catchments of NSW (NSW Industry and Investment, 2009). This thesis identified the top five percent of in stream barriers most critical for restoring stream network connectivity of the Hunter River. These findings could be of value to government and regional authorities, such as the Hunter-Central Rivers Catchment Management Authority (HCRCMA), by providing guidance on areas where future barrier improvements should occur. The findings also accentuate the importance of the main waterways and undeveloped streams to maintaining high levels of connectivity in the Hunter River catchment. Future developments in such regions should be carried out under strict control to ensure that all potential in-stream barriers comply with design standards so as to minimise and ideally prevent negative side-effects. Furthermore, this thesis identified the importance of smaller in-stream barriers for stream network connectivity, such as weirs and road crossings, over the more obvious, commonly targeted and larger dams. Even though weirs were substantially less abundant than dams, they were substantially more important to the restoration of stream network connectivity due to their central location in the landscape. These findings emphasise the need for a shift in both perceptions and action on in-stream barriers and restoring connectivity. Smaller structures can play a major role in fragmenting networks, thus requiring further attention by both management and researchers.

The approach developed in this thesis advocates a technique for informed decision making and as a result is intricately linked and significant to management of numerous river systems. The optimisation model was developed so as to identify the optimal barriers to remove to achieve maximum improvements in connectivity within monetary constraints, which is a significant reality in the implementation of real life projects and, as such, the model is highly useful and valuable to management authorities. The optimisation model is highly robust and can be applied to numerous river systems that are representable by a unidirectional branching network. Additionally, restoration of connectivity can be uniquely evaluated to specifically target certain species and even differing life stages, from juveniles to adult fish. Moreover, the principal unit of restoration for the approach is the catchment, which coincides with the spatial extents over which many management authorities operate. However, the approach is also applicable to sub-catchments, which is potentially useful in larger catchments such as the Murray-Darling system, where it has managerially been divided into 21 regions of which each is being run by an individual catchment management authority (MDBA, 2013).

The findings in this thesis emphasise the effectiveness and flexibility of the approach, although simultaneously caution the means in which it is applied. Substantially suboptimal improvements in connectivity can occur if barriers are selected sequentially without any consideration of future projects. During the project planning phase, numerous restoration scenarios should be analysed to assess whether the selected barrier is actually only achieving a localised optimal improvement. This can prevent misinformed investment of funds and achieve better outcomes.

7.5 LIMITATIONS OF THESIS AND FUTURE RESEARCH

This study has identified potentially critical barriers and regions for restoring longitudinal connectivity to the Hunter River, through a generic optimisation restoration model whose sensitivity was initially tested on the Williams River sub-catchment. This study evaluated connectivity restoration for diadromous and potamodromous types of migration separately to investigate the influence of migration type on restoration outcomes. The findings indicated both differences and similarities in restoration between the two migration types, and were a first step in improving our knowledge on holistic stream network connectivity restoration. However, further research on the simultaneous restoration of both migration types needs to be carried out, as the Hunter River and many Australian freshwater ecosystems are composed of communities with diverse rather than uniform migratory behaviours. The model developed in this thesis can and should be applied for this purpose, by varying the weighting placed on each type of migratory behaviour.

Furthermore, as additional data on the Hunter River catchment is gathered the model can be further refined to improve the quality and accuracy of predictions for the stream network. This is especially important in relation to barrier characteristics, where all structures in the basin were considered as barriers. With further research and fieldwork, fully porous structures will be separable from the actual barriers dataset, and it will be possible to quantify and investigate the extent to which these conservative assumptions impact on restoring connectivity. In addition, habitat quality appeared to have an insignificant impact on connectivity restoration in this thesis, although only one method was applied in its quantification. Further studies investigating a multitude of habitat quality assessment techniques would strengthen current knowledge on the influence of habitat quality on network connectivity restoration.

The findings of this study have been limited to a single study area, even though the approach developed is highly robust. Further research across numerous drainage basins would not only improve understanding of where restorations in each individual basin should be carried out, but also advance general understanding on restoring connectivity and the importance of various riverine landscape attributes in this procedure. Considering that Australian river systems are some of the most hydrologically variable in the world (Puckridge *et al.*, 1998), it is important to gain further understanding on how to account for this variability when optimally restoring stream network connectivity. The model developed in this study can be applied to the biota inhabiting these variable landscapes, which have evolved specific seasonal or flow-cued migratory requirements in response to the regional hydrologic variability. Such analyses that specially target connectivity restoration during migratory events or life-stages can be performed with specific input data on: migratory characteristics of species, their swimming abilities during the migratory life-stage, general flow characteristics and habitat requirements during migration. Investigating the influence of a multitude of species at a number of life-stages on network connectivity restoration could further our understanding on the temporal context of landscape connectivity, which was not investigated in this thesis. Such knowledge would be valuable to management as it could provide general guidelines on the relative importance of various riverine attributes and the influence of target species in restoring connectivity. In depth knowledge on restoring connectivity of entire stream networks appears to be limited to predominantly a number of methodological papers.

The optimisation model developed in this thesis had some limitations, providing opportunities for further research and improvement of the model. Calculation of the DCI was computationally expensive and slow, with the time required compounding as the number of barriers fragmenting a stream network increased. Improvements in calculation time will enable longitudinal connectivity restoration analyses to be carried out at a more rapid pace, thus allowing for more assessments over substantially more fragmented basins.

Furthermore, the optimisation model developed in this thesis is only concerned with restoring longitudinal connectivity of degraded riverine landscapes by barrier improvements; however, these systems have been simultaneously impacted in numerous other ways. Potential for further research could involve improvement and adjustment of the original model so as to incorporate restoration of not only barriers but also patch quality of the stream network. Both of barrier characteristics and habitat quality determine the connectivity of the system, however only restoration of the former is achievable with the current model. Numerous restoration options ranging from in-stream habitat improvement to floodplain reconnection could be incorporated, as they potentially advance the habitat value of the respective stream segment and thus connectivity. Incorporating the option of restoring either patch boundaries or patch quality can pave the way towards achieving more holistic quantitative restoration of riverine landscapes. In addition, such a model would be of paramount value to river management, where it is common for various types of restorations and projects to be implemented simultaneously in a single system. A more holistic model may help achieve optimal improvements in general ecosystem health.

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APPENDIX A – Freshwater and Estuarine Finfish of the Hunter River and Central Rivers Region

No.	Scientific Name	Common Names	Status
Amphidromous Migration Type[1, 5]			
1	Acanthopagrus australis	Yellowfin/silver bream	Common
2	Chanos chanos	Milkfish	Common
3	Galaxias brevipinnis	Climbing galaxias	Uncertain
4	Gobiomorphus australis	Striped gudgeon	Common
5	Leiopotherapon unicolor	Spangled perch	Common
6	Macquaria colonorum	Estuary perch	Uncertain
7	Megalops cyprinoids	Oxeye herring	Common
8	Mugil cephalus	Striped/Sea mullet	Common
9	Myxus elongatus	Sand mullet	Common
10	Platycephalus fuscus	Dusky flathead	Common
11	Pseudomugil signifer	Pacific blue-eye	Common
12	Redigobius macrostoma	Largemouth goby	Common
13	Valamugil georgii	Fantail mullet	Common
Catadromous Migration Type[2, 5]			
14	Anguillia australis	Short-finned eel	Common
15	Anguilla reinhardtii	Long-finned eel	Common
16	Galaxias maculatus	Common jollytail	Common
17	Macquaria novemaculeata	Australian bass	Uncertain
18	Myxus pertardi	Freshwater mullet	Common
19	Notesthes robusta	Bullrout	Limited
20	Potamalosa richmondia	Freshwater herring	Limited
Potamodromous Migration Type[3]			
21	Gobiomorphus coxii	Cox's gudgeon	Common
22	Hypseleotris galii	Firetailed gudgeon	Common
23	Retropinna semoni	Australian smelt	Common
Anadromous Migration Type[4, 5]			
24	Arius graeffei	Freshwater forktailed catfish	Common
Local, Unknown or Migration Type Not listed			
25	Ambassis marianus	Estuary/glass perchlet	Common
26	Atherinosoma microstoma	Smallmouthed hardyhead	Common
27	Afurcagobius tamarensis	Tamar River goby	Common
28	Aldrichetta forsteri	Yellow-eye mullet	Common
29	Amniataba percoides	Banded grunter	Exotic
30	Amoya bifrenatus	Bridled goby	Common
31	Arrhamphus sclerolepis	Snub-nosed garfish	Common
32	Caranx sexfasciatus	Bigeye trevally	Common
33	Bigeye trevally	Goldfish	Exotic
34	Carcharhinus leucas	Bull shark	Common

35	Cyprinus carpio	Common carp	Exotic
36	Elops hawaiensis	Giant herring	Common
37	Epinephelus daemelii	Black cod	Vulnerable
38	Galaxias olidus	Mountain galaxias	Common
39	Gambusia holbrooki	Gambusia, Plague minnow	Exotic
40	Hippichthys penicillus	Steep-nosed pipefish	Common
41	Hypseleotris compressa	Empire gudgeon	Common
42	Hypseleotris klunzingeri	Western carp gudgeon	Common
43	Gerres subfasciatus	Silver biddy	Common
44	Liza argentea	Flat-tail mullet	Common
45	Lutjanus argentimaculatus	Mangrove Jack	Common
46	Monodactylus argenteus	Diamondfish/Silver batfish	Common
47	Mugilogobius platynotus	Flat backed goby	Common
48	Oncorhynchus mykiss	Rainbow trout	Exotic
49	Philypnodon grandiceps	Flathead gudgeon	Common
50	Philypnodon sp.	sp. Dwarf flathead gudgeon	Common
51	Pristis zijsron	Green sawfish	Endangered
52	Pseudogobius sp 9	Blue-spot goby	Common
53	Rhabdosargus sarba	Tarwhine	Common
54	Salvelinus fontinalis	Brook Char	Exotic
55	Salmo trutta	Brown trout	Exotic
56	Scatophagus argus	Spotted scat	Common
57	Selenotoca multifasciata	Banded scat	Common
58	Tandanus tandanus	Eel tail catfish	Common
59	Terapon jarbua	Crescent Perch	Common

Source: NSW DPI (2006b)

[1] Amphidromous – fish that migrate between the ocean and freshwater but not for spawning purposes

[2] Catadromous – fish that predominantly live in freshwater but migrate to the ocean to spawn

[3] Potamodromous – fish that migrate only within freshwater

[4] Anadromous – fish that predominantly live in the ocean but migrate to freshwater to spawn

[5] Amphidromous, catadromous and anadromous migration types fall under the diadromous migration type, which incorporates all species that carry out migrations between the ocean and freshwater (McDowall, 2007)

APPENDIX B – Critical Barriers of the Williams River Network for Diadromous and Potamodromous Migration Type

Waterway	Street/Barrier Name	Barrier Type	Location		% Times Selected	
			Latitude	Longitude	Diadromous[1]	Potamodromous[2]
Williams River	East-Seaham Rd	Road Crossing	-32.662699	151.732645	100	-
Williams River	Limeburners Creek Rd	Road Crossing	-32.581181	151.782196	100	100
Williams River	Pine-Brush Rd	Road Crossing	-32.520361	151.797858	100	100
Williams River	Seaham Weir	Weir	-32.66041	151.737454	100	-
Williams River	Alison Rd	Road Crossing	-32.468148	151.759099	98.8	100
Balickera Canal	East-Seaham Rd	Road Crossing	-32.659881	151.742748	98.5	-
Williams River	Fosterton Rd	Road Crossing	-32.324985	151.747755	95	98.8
Williams River	Stroud Hill Rd	Road Crossing	-32.397393	151.76425	95	98.8
Williams River	Rail Line to Dungog	Rail Crossing	-32.396679	151.763431	95	98.8
Williams River	Chichester Dam Rd	Road Crossing	-32.300722	151.717032	-	95.9
Williams River	Bandon Grove Weir	Weir	-32.301631	151.716269	-	95.9

[1] List of critical barriers for diadromous migration type (Figure 5.5), and proportion of time each barrier was selected to be modified

[2] List of critical barriers for potamodromous migration type (Figure 5.9), and proportion of time each barrier was selected to be modified

APPENDIX C – Critical Barriers of the Hunter River Network (Fine Scale of Restoration: Top 25)

Waterway	Street/Barrier Name	Barrier Type	Location		Class	
			Latitude	Longitude	Diadromous[1]	Potamodromous[2]
Hunter River	Teal St/Nelson Bay Rd	Road Crossing	-32.883806	151.786871	1	-
Hunter River	Pacific Hwy from Hexham	Road Crossing	-32.823027	151.685422	1	-
Hunter River	Raymond Terrace Rd	Road Crossing	-32.751162	151.726453	0.8	-
Hunter River	Phoenix Park Rd	Road Crossing	-32.723852	151.626607	0.8	-
Hunter River	Belmore Rd	Road Crossing	-32.728612	151.5535	0.8	-
Hunter River	Rail Line to Oakhampton Rd	Rail Crossing	-32.694751	151.568472	0.8	-
Hunter River	Aberglasslyn Rd	Road Crossing	-32.688776	151.525165	0.8	-
Hunter River	Luskintyre Rd	Road Crossing	-32.685725	151.431442	0.8	-
Hunter River	Derelict Instream Structure	Road Crossing	-32.641131	151.425316	0.8	-
Hunter River	Elderslie Rd	Road Crossing	-32.610548	151.346583	0.8	-
Williams River	Seaham Rd	Road Crossing	-32.753356	151.744752	0.6	-
Hunter River	Queen St/Gresford Rd	Road Crossing	-32.559561	151.195981	0.6	-
Hunter River	New England Hwy	Road Crossing	-32.557842	151.172555	0.6	-
Hunter River	Dunolly Rd	Road Crossing	-32.559526	151.170972	0.6	-
Hunter River	Rail Line by Rose Point Rd	Rail Crossing	-32.563173	151.158994	0.6	-
Williams River	East-Seaham Rd	Road Crossing	-32.662699	151.732645	0.4	-
Williams River	Seaham Weir	Weir	-32.66041	151.737454	0.4	-
Williams River	Limeburners Creek Rd	Road Crossing	-32.581181	151.782196	0.2	-
Williams River	Pine Brush Rd	Road Crossing	-32.520361	151.797858	0.2	-
Williams River	Alison Rd	Road Crossing	-32.468148	151.759099	0.2	-
Hunter River	Private Rd (Mines) off Lemington Rd	Road Crossing	-32.516288	150.989323	0.2	-
Hunter River	Lemington Rd	Road Crossing	-32.51645	150.936566	0.2	-
Hunter River	Jerrys Plains Weir	Weir	-32.489626	150.923393	0.2	-
Hunter River	Liddell Gauging Station	Weir	-32.472194	150.894978	0.2	-
Hunter River	Golden Hwy by Edderton Rd	Road Crossing	-32.452911	150.849666	0.2	-
Hunter River	Unnamed Rd off Bureen Rd	Road Crossing	-32.472483	150.78729	0.2	-
Hunter River (Upper)	Golden Hwy from Denman	Road Crossing	-32.378996	150.711188	0.2	0.6

Goulburn River	Martindale Rd	Road Crossing	-32.431934	150.672949	0.2	0.6
Goulburn River	Yarrawa Rd	Road Crossing	-32.412214	150.64054	0.2	0.6
Goulburn River	Bylong Valley Way	Road Crossing	-32.346015	150.573185	0.2	0.6
Goulburn River	Rail Line north of Bylong Valley Way	Rail Crossing	-32.402369	150.375771	0.2	0.8
Goulburn River	Rail Line north of Wollar Rd	Rail Crossing	-32.344768	150.065063	-	1
Goulburn River	Rail Line north of Wollar Rd	Rail Crossing	-32.341941	150.048559	-	1
Goulburn River	Wollar Rd	Road Crossing	-32.32795	150.037453	-	1
Goulburn River	Private Rd off Hulks Rd	Road Crossing	-32.21695	150.061711	-	1
Munmurra River	Private Rd off Summer Hill Rd	Road Crossing	-32.149599	150.00918	-	1
Krui River	Golden Hwy to Cassilis	Road Crossing	-32.095456	150.118115	-	0.8
Widden Brook	Bylong Valley Way	Road Crossing	-32.405208	150.373566	-	0.8
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.426894	150.38243	-	0.8
Widden Brook	Emu Creek Rd	Road Crossing	-32.519663	150.362759	-	0.8
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.548852	150.353767	-	0.4
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.553412	150.359639	-	0.4
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.576747	150.373525	-	0.4
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.615564	150.364686	-	0.4
Blackwater Creek	Widden Valley Rd	Road Crossing	-32.635042	150.368301	-	0.4
Four Mile Creek	Golden Hwy by Cassilis	Road Crossing	-32.028529	149.954055	-	0.4
Bylong River	Rail Line alongside Wollar Rd	Rail Crossing	-32.350049	150.09334	-	0.2
Bylong River	Wollar Rd	Road Crossing	-32.351096	150.092606	-	0.2
Bylong River	Private Rd off Wollar Rd	Road Crossing	-32.363768	150.115592	-	0.2
Bylong River	Bylong Valley Way	Road Crossing	-32.411728	150.113741	-	0.2
Bylong River	Upper Bylong Rd	Road Crossing	-32.433541	150.13453	-	0.2
Bylong River	Private Rd off Upper Bylong Rd	Road Crossing	-32.452199	150.178536	-	0.2

[1] List of critical barriers for diadromous migration type (Figure 6.2a) and their class (Figure 6.1; Table 6.2)

[2] List of critical barriers for potamodromous migration type (Figure 6.2b) and their class (Figure 6.1; Table 6.2)

APPENDIX D – Critical Barriers of the Hunter River Network (Coarse Scale of Restoration: Top 125)

Waterway	Street/Barrier Name	Barrier Type	Location		Class	
			Latitude	Longitude	Diadromous[1]	Potamodromous[2]
Hunter River	Teal St/Nelson Bay Rd	Road Crossing	-32.883806	151.786871	1	-
Hunter River	Pacific Hwy from Hexham	Road Crossing	-32.823027	151.685422	1	0.6
Hunter River	Raymond Terrace Rd	Road Crossing	-32.751162	151.726453	1	0.8
Hunter River	Phoenix Park Rd	Road Crossing	-32.723852	151.626607	1	0.8
Hunter River	Belmore Rd	Road Crossing	-32.728612	151.5535	1	0.8
Hunter River	Rail Line to Oakhampton Rd	Rail Crossing	-32.694751	151.568472	1	0.8
Hunter River	Aberglasslyn Rd	Road Crossing	-32.688776	151.525165	1	0.8
Hunter River	Luskintyre Rd	Road Crossing	-32.685725	151.431442	1	0.8
Hunter River	Derelict Instream Structure	Road Crossing	-32.641131	151.425316	1	0.8
Hunter River	Elderslie Rd	Road Crossing	-32.610548	151.346583	1	0.8
Hunter River	Queen St/Gresford Rd	Road Crossing	-32.559561	151.195981	1	0.8
Hunter River	New England Hwy	Road Crossing	-32.557842	151.172555	1	0.8
Hunter River	Dunolly Rd	Road Crossing	-32.559526	151.170972	1	0.8
Hunter River	Rail Line by Rose Point Rd	Rail Crossing	-32.563173	151.158994	1	0.8
Hunter River	Private Rd (Mines) off Lemington Rd	Road Crossing	-32.516288	150.989323	1	0.8
Hunter River	Lemington Rd	Road Crossing	-32.51645	150.936566	1	0.8
Hunter River	Jerrys Plains Weir	Weir	-32.489626	150.923393	1	0.8
Hunter River	Liddell Gauging Station	Weir	-32.472194	150.894978	1	0.8
Hunter River	Golden Hwy by Edderton Rd	Road Crossing	-32.452911	150.849666	1	0.8
Hunter River	Unnamed Rd off Bureen Rd	Road Crossing	-32.472483	150.78729	1	0.8
Hunter River (Upper)	Golden Hwy from Denman	Road Crossing	-32.378996	150.711188	1	1
Goulburn River	Martindale Rd	Road Crossing	-32.431934	150.672949	1	1
Goulburn River	Yarrawa Rd	Road Crossing	-32.412214	150.64054	1	1
Goulburn River	Bylong Valley Way	Road Crossing	-32.346015	150.573185	1	1
Goulburn River	Rail Line north of Bylong Valley Way	Rail Crossing	-32.402369	150.375771	1	1
Goulburn River	Rail Line north of Wollar Rd	Rail Crossing	-32.344768	150.065063	0.8	1
Goulburn River	Rail Line north of Wollar Rd	Rail Crossing	-32.341941	150.048559	0.8	1

Goulburn River	Wollar Rd	Road Crossing	-32.32795	150.037453	0.8	1
Goulburn River	Private Rd off Hulks Rd	Road Crossing	-32.21695	150.061711	0.8	1
Munmurra River	Private Rd off Summer Hill Rd	Road Crossing	-32.149599	150.00918	0.8	1
Krui River	Golden Hwy to Cassilis	Road Crossing	-32.095456	150.118115	0.8	1
Widden Brook	Bylong Valley Way	Road Crossing	-32.405208	150.373566	0.8	1
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.426894	150.38243	0.8	1
Widden Brook	Emu Creek Rd	Road Crossing	-32.519663	150.362759	0.8	1
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.548852	150.353767	0.8	1
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.553412	150.359639	0.8	1
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.576747	150.373525	0.8	1
Widden Brook	Private Rd off Widden Valley Rd	Road Crossing	-32.615564	150.364686	0.8	1
Blackwater Creek	Widden Valley Rd	Road Crossing	-32.635042	150.368301	0.8	1
Four Mile Creek	Golden Hwy by Cassilis	Road Crossing	-32.028529	149.954055	0.8	0.8
Bylong River	Rail Line alongside Wollar Rd	Rail Crossing	-32.350049	150.09334	0.8	1
Bylong River	Wollar Rd	Road Crossing	-32.351096	150.092606	0.8	1
Bylong River	Private Rd off Wollar Rd	Road Crossing	-32.363768	150.115592	0.8	1
Bylong River	Bylong Valley Way	Road Crossing	-32.411728	150.113741	0.8	1
Bylong River	Upper Bylong Rd	Road Crossing	-32.433541	150.13453	0.8	1
Bylong River	Private Rd off Upper Bylong Rd	Road Crossing	-32.452199	150.178536	0.8	1
Merriwa River	Private Rd off Cullingral Rd	Road Crossing	-32.32639	150.34315	0.8	0.8
Cousins Creek	Private Rd off Upper Bylong Road	Road Crossing	-32.457829	150.164773	0.8	0.8
Murrumbline Creek	Durridgerie Rd	Road Crossing	-32.181294	149.854306	0.8	0.6
Borambil Creek	Private Rd off Wyoming Rd	Road Crossing	-32.090237	150.000382	0.8	0.6
Williams River	Seaham Rd	Road Crossing	-32.753356	151.744752	0.6	0.8
Williams River	East-Seaham Rd	Road Crossing	-32.662699	151.732645	0.6	0.8
Williams River	Seaham Weir	Weir	-32.66041	151.737454	0.6	0.8
Williams River	Limeburners Creek Rd	Road Crossing	-32.581181	151.782196	0.6	0.8
Williams River	Pine Brush Rd	Road Crossing	-32.520361	151.797858	0.6	0.6
Williams River	Alison Rd	Road Crossing	-32.468148	151.759099	0.6	0.6
Table Bay Creek	Private Rd off Widden Valley Rd	Road Crossing	-32.675703	150.352218	0.6	0.6
Red Creek	Private Rd off Widden Valley Rd	Road Crossing	-32.71013	150.330781	0.6	0.6
Lee Creek	Private Rd off Upper Bylong Rd	Road Crossing	-32.473755	150.131137	0.6	0.6
Lee Creek	Budden Gap Rd	Road Crossing	-32.48365	150.129351	0.6	0.6
Lee Creek	Private Rd off Upper Bylong Rd	Road Crossing	-32.535098	150.119317	0.6	0.6

Wollombi Brook	Private Rd off Comleroi Rd (Mines)	Road Crossing	-32.554597	151.03664	0.6	0.6
Wollombi Brook	Private Rd off Comleroi Rd (Mines)	Road Crossing	-32.554583	151.028688	0.6	0.6
Wollombi Brook	Golden Hwy at Warkworth	Road Crossing	-32.566354	151.021718	0.6	0.6
Wollombi Brook	Private Rd off Golden Hwy (Mines)	Road Crossing	-32.577588	151.017755	0.6	0.6
Wollombi Brook	Putty Rd	Road Crossing	-32.650654	151.020247	0.6	0.6
Wollombi Brook	Private Rd off Milbrodale Rd	Road Crossing	-32.708314	151.052448	0.6	0.6
Wollombi Brook	Milbrodale Rd	Road Crossing	-32.750228	151.100158	0.6	0.6
Drews Creek	Unnamed Rd off Adams Peak Rd	Road Crossing	-32.790598	151.062396	0.6	0.6
Werong Creek	Private Rd off Unnamed Rd off Stockyard Creek Track	Road Crossing	-32.864185	151.060466	0.6	0.6
Cody Creek	Private Rd off Unnamed Rd off Stockyard Creek Track	Road Crossing	-32.864228	151.040739	0.6	0.6
Werong Creek	Private Rd off Unnamed Rd off Stockyard Creek Track	Road Crossing	-32.869908	151.050251	0.6	0.6
Werong Creek	Private Rd off Unnamed Rd off Stockyard Creek Track	Road Crossing	-32.875747	151.056229	0.6	0.6
Werong Creek	Private Rd off Unnamed Rd off Stockyard Creek Track	Road Crossing	-32.880112	151.056266	0.6	0.6
Doyles Creek	Unnamed Rd of Bureen Rd	Road Crossing	-32.48185	150.806427	0.4	0.4
Doyles Creek	Bureen Rd	Road Crossing	-32.503854	150.801835	0.4	0.4
Doyles Creek	Private Rd off Doyles Creek Rd	Road Crossing	-32.518612	150.795838	0.4	0.4
Doyles Creek	Woodland Hill Track	Road Crossing	-32.544948	150.790632	0.4	0.4
Doyles Creek	Private Rd off Doyles Creek Rd	Road Crossing	-32.55222	150.787419	0.4	0.4
Doyles Creek	Private Rd off Doyles Creek Rd	Road Crossing	-32.553612	150.789128	0.4	0.4
Hunter River (Upper)	Bengalla Rd	Road Crossing	-32.289756	150.845832	0.4	0.4
Hunter River (Upper)	Rail Line by Muswellbrook	Rail Crossing	-32.265981	150.884247	0.4	0.4
Hunter River (Upper)	Kayuga Rd	Road Crossing	-32.255979	150.888054	0.4	0.4
Hunter River (Upper)	Unnamed Rd off Kayuga Rd	Road Crossing	-32.194314	150.875409	0.4	0.4
Hunter River (Upper)	New England Hwy north of Aberdeen	Road Crossing	-32.156395	150.885215	0.4	0.4
Hunter River (Upper)	Rail Line north of Aberdeen	Rail Crossing	-32.1545	150.888867	0.4	0.4
Hunter River (Upper)	Glenbawn Rd	Road Crossing	-32.136571	150.959839	0.4	0.4
Hunter River (Upper)	Unnamed Rd off Rouchel Rd	Road Crossing	-32.143458	150.975546	0.4	0.4
Hunter River (Upper)	Unnamed Rd off Glenbawn Rd	Road Crossing	-32.118493	150.983962	0.4	0.4
Hunter River (Upper)	Glenbawn Dam Weir	Weir	-32.111958	150.991021	0.4	0.4
Hunter River (Upper)	Glenbawn Dam	Dam	-32.109271	150.998253	0.4	0.4
Rouchel Brook	Dangarfield Rd	Road Crossing	-32.137152	150.997496	0.4	0.4
Rouchel Brook	Private Rd off Rouchel Rd	Road Crossing	-32.148027	151.062621	0.4	0.4
Dart Brook	Unnamed Rd off Kayuga Rd	Road Crossing	-32.196202	150.872178	0.4	0.4
Dart Brook	Unnamed Rd off Blairmore Rd	Road Crossing	-32.191472	150.864084	0.4	0.4

Dart Brook	Blairmore Rd	Road Crossing	-32.188665	150.861426	0.4	0.4
Dart Brook	Dartbrook Rd	Road Crossing	-32.144549	150.854075	0.4	0.4
Kingdon Ponds	Dartbrook Rd	Road Crossing	-32.145318	150.868052	0.4	0.4
Kingdon Ponds	Turanville Rd	Road Crossing	-32.101755	150.855299	0.4	0.4
Kingdon Ponds	Liverpool St	Road Crossing	-32.048042	150.851235	0.4	0.4
Cutroad Arm Creek	Private Rd off Unnamed Rd off Stockyard Creek Track	Road Crossing	-32.870055	151.049613	0.2	0.2
Parsons Creek	Unnamed Rd off Putty Rd	Road Crossing	-32.680411	151.026618	0.2	0.2
Parsons Creek	Unnamed Rd off Putty Rd	Road Crossing	-32.68596	151.019667	0.2	0.2
Parsons Creek	Unnamed Rd off Putty Rd	Road Crossing	-32.685793	151.015223	0.2	0.2
Parsons Creek	Putty Rd	Road Crossing	-32.685212	151.010831	0.2	0.2
Milbrodale Creek	Private Rd off Thompsons Rd	Road Crossing	-32.684369	151.00571	0.2	0.2
Milbrodale Creek	Private Rd off Thompsons Rd	Road Crossing	-32.682343	150.992852	0.2	0.2
Parsons Creek	Private Rd off Thompsons Rd	Road Crossing	-32.685219	151.006067	0.2	0.2
Parsons Creek	Private Rd off Putty Rd	Road Crossing	-32.697453	150.97184	0.2	0.2
Dingo Creek	Bylong Valley Way	Road Crossing	-32.381147	150.439642	0.2	0.2
Myrtle Creek	Myrtle Gully Rd	Road Crossing	-32.525538	150.34685	0.2	0.2
Myrtle Creek	Private Rd off Myrtle Gully Rd	Road Crossing	-32.538254	150.280657	0.2	0.2
Turon Creek	Emu Creek Rd	Road Crossing	-32.56293	150.306443	0.2	0.6
Wilpinjong Creek	Private Rd off Ulan Wollar Rd	Road Crossing	-32.314895	149.932128	0.2	0.2
Horse Creek	Private Rd off Cullingral Rd	Road Crossing	-32.322805	150.347159	0.2	0.2
Merriwa River	Private Rd off Cullingral Rd	Road Crossing	-32.284739	150.328908	0.2	0.2
Merriwa River	Private Rd off Cullingral Rd	Road Crossing	-32.25268	150.326461	0.2	0.2
Merriwa River	Private Rd off Cullingral Rd	Road Crossing	-32.234169	150.324209	0.2	0.2
Farm Springs Creek	Cullingral Rd	Road Crossing	-32.21295	150.318306	0.2	0.2
Merriwa River	Golden Hwy by Merriwa	Road Crossing	-32.138664	150.348846	0.2	0.2
Merriwa River	Private Rd off Blaxland St	Road Crossing	-32.136089	150.351447	0.2	0.2
Merriwa River	Mountain Station Rd	Road Crossing	-32.094831	150.363886	0.2	0.2
Coulsons Creek	Private Rd off Willow Tree Rd	Road Crossing	-32.086309	150.370335	0.2	0.2
Mountain Station Creek	Mountain Station Rd	Road Crossing	-32.001931	150.350468	0.2	0.2
Gummum Creek	Gummum Rd	Road Crossing	-31.9905	150.355037	0.2	0.2
Cream of Tartar Creek	Cream of Tartar Rd	Road Crossing	-31.964496	150.344299	0.2	0.2
Wilpinjong Creek	Unnamed Rd off Ulan Wollar Rd	Road Crossing	-32.310982	149.855234	-	0.2

[1] List of critical barriers for diadromous migration type (Figure 6.4a) and their class (Figure 6.3; Table 6.3)

[2] List of critical barriers for potamodromous migration type (Figure 6.4b) and their class (Figure 6.3; Table 6.3)