SPATIAL OPTIMISATION FOR RIVER RESTORATION PLANNING IN NOVA SCOTIA, CANADA

by

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DEDICATION

I would like to dedicate this thesis to my parents, Mary and Doug. Thank you for always being patient, kind, and supportive.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

River restoration is believed to have the greatest chance of success when action is considered in the broader context of the riverscape. However, methods are lacking to fully integrate systemic connectivity into decision-making. Optimisation, a method of prioritisation, is capable of accounting for longitudinal connectivity, spatial interdependence, and cumulative effects of anthropogenic barriers such as dams and culverts. In addition, optimisation can help ensure that limited restoration funds are efficiently allocated. Despite these advantages, it remains under-employed. I present optimisation models for maximising connectivity within a river network (i.e., undirected connectivity) and connectivity between the network and its outflow (i.e., directed connectivity) and demonstrate their application on three river networks in Nova Scotia, Canada. Non-additive cumulative effects of barriers and key budget thresholds that yielded better returns on investment were observed. The methods and models address current challenges in implementation of the optimisation approach to systematic river restoration planning.

LIST OF ABBREVIATIONS USED

COSEWIC	Committee on the Status of Endangered Wildlife in Canada
DCI	Dendritic Connectivity Index
DCI_d	Diadromous Dendritic Connectivity Index
DCI_p	Potadromous Dendritic Connectivity Index
DEN	Dendritic Ecological Network
DSS	Decision Support System
ESRI	Environmental Systems Research Institute
FPB	Forest Practices Board
GIS	Geographic Information Systems
GLPK	GNU Linear Programming Kit
GWh	Gigawatt-hours
HIS	Habitat Suitability Index
ILP	Integer Linear Programming
IP	Integer Programming
LP	Linear Programming
MCDM	Multi-Criteria Decision-Making
MILP	Mixed Integer Linear Programming
MW	Megawatts
NSHN	Nova Scotia Hydrographic Network
NSPI	Nova Scotia Power Incorporated
NSTDB	Nova Scotia Topographic Database
SDSS	Spatial Decision Support System
SR	Scoring and Ranking
$ZMAX_d$	Maximum Permeability-Weighted Accessible Directed Network
$ZMAX_u$	Maximum Permeability-Weighted Accessible Undirected Network

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CHAPTER 1: INTRODUCTION

1.1 General Introduction & Problem Summary

Human activity has led to a severe decline of global freshwater biodiversity (Dudgeon et al., 2006; Vörösmarty et al., 2010), even more so than in other biomes (Strayer & Dudgeon, 2010). Anthropogenic fragmentation of river systems has occurred on a global scale, in large part due to damming (Dynesius & Nilsson, 1994; World Commission on Dams, 2000; Nilsson et al., 2005; Lehner et al., 2011), and is believed to be a significant contributor to the rapid and widespread decline of freshwater and freshwater-marine migratory fish (Pringle et al., 2000; Dudgeon et al., 2006; Greathouse et al., 2006; Limburg & Waldman, 2009; Humphries & Winemiller, 2009; Vörösmarty et al., 2010; Moyle et al., 2011; Horreo et al., 2011; Rolls, 2011; Liermann et al., 2012). Smaller barriers, such as those commonly found at road crossings, are also known to act as ecological stressors (Wheeler et al., 2005; Park et al., 2008; Eberhardt et al., 2011; Conesa-García & García-Lorenzo, 2012; Eggert, 2012), impairing fish movement to the detriment of fish assemblages (Belford & Gould, 1989; Vander Pluym et al., 2008; Alexandre & Almeida, 2010; Nislow et al., 2011; MacPherson et al., 2012). In response, extensive restoration efforts have been undertaken to mitigate the impact of anthropogenic river barriers such as dams, weirs, and culverts (for a review see Bernhardt et al., 2005).

Prioritisation of river restoration projects must be undertaken because of resource limitations (i.e., time, labour, capital) and the large number of barriers present on typical river systems. Though it remains infrequently used, optimisation has been shown to select more efficient priorities than more common methods of prioritisation, such as scoring and ranking (SR; O'Hanley & Tomberlin, 2005; O'Hanley et al., 2013). In addition, it is now widely recognised that any systematic prioritisation should consider expected ecological benefits at the riverscape scale (Giller, 2005; Jansson et al., 2007; Lake et al., 2007; Palmer, 2009; Beechie et al., 2010). One method is to quantify benefits to systemic connectivity along the length of a river from headwaters to outflow, or *longitudinal* *connectivity* (Ward, 1989; Cote et al., 2009), though this presents challenges (see Fullerton et al., 2010). Notable recent advances have been made through the adoption and application of conceptual frameworks that emphasise the underlying network structure of rivers (Benda et al., 2004; Proulx et al., 2005; Erős et al., 2011), envision rivers as *dendritic ecological networks* (DENs; Fagan, 2002; Grant et al., 2007; Cote et al., 2009; Peterson et al., 2013), and draw on techniques from network theory (e.g., Dale & Fortin, 2010; Erős et al., 2012; Segurado et al., 2013). The dendritic connectivity index (DCI), for example, has facilitated empirical study of longitudinal connectivity (Cote et al., 2009). However, instances of restoration prioritisation that adequately integrate systemic connectivity into decision-making are still rare (Lake et al., 2007; Roni et al., 2008; Beechie et al., 2008).

In systematic river restoration, there are two ecologically relevant sub-classifications of longitudinal connectivity that are important to consider. The first type, herein referred to as *directed* connectivity, is the degree to which upper reaches of the system are connected to the outflow, or *sink*, and vice versa (O'Hanley & Tomberlin, 2005; Cote et al., 2009). Directed connectivity is crucial to migration and spawning of marine-freshwater migratory (i.e., *diadromous*) fish (Peter, 1998; Larinier, 2000; Katano et al., 2006; Morita et al., 2009; Smith & Hightower, 2012) and to the transport of nutrients, woody debris, and sediment (e.g., Kroeze et al., 2012). The second type of longitudinal connectivity, herein referred to as *undirected connectivity,* is the degree of connectivity between any given point in the river system and all other points in the system, regardless of the direction of flow (Cote et al., 2009; O'Hanley, 2011). Some freshwater species, such as resident migratory (i.e., *potamodromous*) fish, migrate along the fluvial length of river networks with relatively little directional bias (Warren & Pardew, 1998; Lamphere & Blum, 2012) and thus require a distinctly undirected type of connectivity. Loss of undirected connectivity restricts the movement and adversely affects populations of resident fish (Warren & Pardew, 1998; Porto et al., 1999; Nislow et al., 2011; Perkin & Gido 2012) and can lead to local extirpations (Winston et al., 1991; Tsuboi et al., 2010). The unusually strong connectivity of riverscapes compared to other landscapes is now widely recognized as fundamental to their structure and function (Melles et al., 2012).

The development and application of models and methodological frameworks that satisfactorily incorporate both directed and undirected connectivity are therefore needed (Lake et al., 2007; Schick & Lindley, 2007; Moilanen et al., 2008).

Although optimisation models exist for maximising both directed (e.g., Kuby et al., 2005; O'Hanley & Tomberlin, 2005; Zheng et al., 2009) and undirected connectivity (e.g., O'Hanley, 2011; O'Hanley et al., 2013), examples of optimisation used in river barrier prioritisation are still uncommon. Inflexibility of existing models, lack of transparency to decision-makers, high computational burden, high cost of software and expertise required, general underexposure, and lack of understanding of optimisation are all factors believed to be contributing to the slow uptake of the method (O'Hanley & Tomberlin, 2005; Beechie et al., 2008). In addition, the effort required to visualise results on a map is likely also a contributing factor (Beechie et al., 2008). Geographic information systems (GIS) have shown particular promise in facilitating the practical application of ecological and network theory to river restoration (e.g., Poplar-Jeffers et al., 2009; Zheng et al., 2009; Kemp & O'Hanley, 2010; Mount et al., 2011) but, to my knowledge, have not yet been used in concert with optimisation in systematic river restoration planning. A software package or spatial decision support system (SDSS; Malczewski, 1999), similar to those developed and frequently applied in systematic freshwater *conservation* planning (Margules & Pressey, 2000; Moilanen et al., 2008; Moilanen et al., 2009; Pressey et al., 2009; Watts et al., 2009; Linke et al., 2011; Turak & Linke, 2011), will likely be necessary to make optimisation modeling more commonplace in this context.

Regardless of the prioritisation method chosen, data acquisition and model parameterisation pose a challenge for systematic river restoration planning. Estimating segment size, river network quality or habitat suitability, and barrier permeability (i.e., passability) often consumes considerable time and resources. Stream size is recognised to have a number of dimensions: wetted width, flow discharge, mean depth, time of year, and number of contributing tributaries (see Hughes et al., 2010, for a review). The quality of river network, in a biological sense, can be expressed using such methods as habitat suitability indices (HSI, e.g., Kocovsky et al., 2009), measures of ecological health (e.g., Mader & Maier, 2008; Zheng et al., 2009), dispersal models (e.g., Schick & Lindley, 2007; Pépino et al., 2012; Muehlbauer et al., 2013), and species presence-absence data (e.g., Mount et al., 2011; G. Anderson et al., 2012). Length is prevalently used to quantify river network size (O'Hanley & Tomberlin, 2005; Hicks & Sullivan, 2008; Mader & Maier, 2008; Kocovsky et al., 2009; Mount et al., 2011; Anderson et al., 2012; Nunn & Cowx, 2012; Fish Passage Technical Working Group, 2012), despite surface area being a better representation of habitat size from the perspective of aquatic organisms (Hughes et al., 2010; Cote et al., 2011) and more appropriate to the study of stream-lake networks (see Rosenfeld & Jones, 2010). Presumably, the frequent choice of length is due to the scarcity of surface area data at the broad scale for smaller network features, such as lower order streams. Barrier permeability reflects the degree to which single barriers can affect longitudinal connectivity (Kemp & O'Hanley, 2010; Bourne et al., 2011). Its various dimensions include direction, timing, active versus passive movement (e.g., fish populations versus organic matter), species, and physiology and fitness of individuals within a population (for reviews see Kemp & O'Hanley, 2010; Bourne et al., 2011; Anderson et al., 2012). Bourne *et al.* (2011) found that SR prioritisations were insensitive to barrier permeability estimates, though it remains unknown how sensitive prioritisations identified by optimisation are to permeability. With respect to these three parameters in particular, understanding how data accuracy influences connectivity assessments and subsequent prioritisations is therefore important.

The presence of various types of barriers on river networks augments the difficulty of data acquisition, connectivity impact assessment, and barrier prioritisation. Hydropower dams typically occur on larger river segments and are costly to mitigate. Improperly installed culverts (pipes found under roads at stream crossings) are a second type that are usually found on streams or smaller river segments. While they are less costly to repair, they are far more numerous than dams (Januchowski-Hartley et al., 2013). The detrimental effects of dams on longitudinal connectivity are well-established (e.g., Humphries & Winemiller, 2009), yet there are indications that the cumulative effects of 'small' barriers are also significant (Beechie et al., 1994; Poplar-Jeffers et al., 2009; Alexandre & Almeida, 2010; Januchowski-Hartley et al., 2013). The trade-offs between restoration of longitudinal connectivity at hydropower dams and smaller, less costly barriers remain difficult to assess, with spatial interdependence and vast numbers of barriers contributing to the challenge (Palmer, 2009; Kemp & O'Hanley, 2010). Past studies have examined effects of small obstacles versus larger dams on fish populations (e.g., Beechie et al., 1994; Alexandre & Almeida, 2010), though there have been few studies which have explicitly examined the cumulative effects of these barriers (though see Januchowski-Hartley et al., 2013) or conducted optimisation analysis to prioritise their mitigation.

The importance of considering cumulative effects of restoration projects at the watershed scale has been heavily emphasized (Palmer & Bernhardt, 2006; Jansson et al., 2007; Palmer, 2009; Poplar-Jeffers et al., 2009; Beechie et al., 2010). However, there is ambiguity surrounding the definition of this term (see Duinker et al., 2013 for a review). Herein, the study of cumulative effects shall be accepted to encompass the *removal of a single ecological stressor*, focusing on whether ecological responses are simple or complex. Generally, *simple* cumulative effects are defined as those that are additive, whereas *complex* cumulative effects are non-additive, displaying synergism or antagonism - that is, interactions occur that contribute effects greater (i.e., synergistic) or less (i.e., antagonistic) than the expected total (Folt et al., 1999; Diefenderfer et al., 2011).

The degree to which longitudinal connectivity restoration projects exhibit non-additive effects is relevant to systemic restoration planning because SR and step-wise SR are less effective than optimisation for arriving at efficient prioritisations when restoration projects display non-additive cumulative effects. This is due to the problem of considering combinations of restoration projects. Synergism (i.e., non-additive cumulative effects) of barrier mitigation can be reflected by a positive connectivity-gain-to-budget relationship (i.e., *economies of scale*; see Diefenderfer et al., 2012) observed in the results of optimisation. Another indicator of synergisms arising from interdependence of restoration actions are occasions where options that appeared in optimised priorities at a lower budget increment do not appear at a higher one, referred to as *non-nestedness*.

Both non-nestedness and a positive connectivity-gain-to-budget relationship can arise due to budget increases correspondingly increasing the potential for synergistic interactions between barrier mitigation projects. However, variable project costs can also explain such observations; certain high-return projects may not be affordable until budget thresholds are reached. Thus, to isolate synergistic cumulative effects, cost must be controlled for. This can be achieved by weighting all projects equally (i.e., using prioritised project count as surrogate 'cost' constraint) in optimisation.Insights into the type of systemic response to restoration of directed and undirected longitudinal connectivity can therefore be garnered through analysing optimisation results and conducting additional cost-controlled optimisation analyses.

1.2 Research Objectives

The following specific research objectives (numbered) and related questions (lettered) were identified:

1) Develop and apply optimisation models for maximising both directed and undirected longitudinal connectivity.
    a. How can optimisation models be formulated in ways that remain flexible to data availability and various decision-making scenarios, have reasonably low computational burden, minimize the expense of software and expertise required to run analyses, and account for both the directed and undirected ecological characteristics of river networks?
2) Couple optimisation models with a SDSS and propose methods to help overcome issues preventing more-common usage of optimisation in river restoration planning.
    b. What advantages or limitations are there to embedding optimisation models within a GIS-based SDSS?
3) Demonstrate these models on real-world systems.
    c. Can the models developed solve realistic prioritisation problems?
    d. Do estimates of longitudinal connectivity vary between different methods of quantifying network size?
    e. For river systems such as these containing hydropower dams, does the observed relationship between budget and connectivity gains conform to those observed in past studies? Why or why not?
    f. How important is accurate assessment of culvert permeability? How is imperfect information likely to affect the outcomes of restoration on these river systems?

g. Do the cumulative effects of relatively numerous culverts coupled with the low cost of their mitigation make them a more efficient choice than hydropower dams, which are relatively expensive to mitigate?
h. How much do culverts as a group compare to dams as a group in terms of effects to systemic connectivity?
i. Are restoration efforts likely to exhibit additive or non-additive (i.e., synergistic) effects on these systems?

1.3 Project Overview

In this thesis, I present two linear optimisation models for maximising directed and undirected longitudinal connectivity given a limited budget. These models are flexible in that they are solvable using common, relatively low-cost (or free) software, have low minimal data requirements, and incorporate continuous barrier permeability values. They also can accommodate optimisation objectives that aim to maximise a recently developed systemic measure of connectivity, the DCI (Cote et al., 2009), or, alternatively, absolute measures of river network size (e.g., river length or surface area). Analyses were conducted using GIS software and a customisation of an existing SDSS, integrated with off-the-shelf optimisation software. I applied these models to analyses of three river systems in Nova Scotia, all of which have both hydropower dams and numerous culverts present. The intention of this research was not to prescribe specific mitigation measures for the three study systems, given that the verification of all barrier locations, project costs, and barrier permeabilities was beyond the scope of the study. Rather, the realistic topology of these networks and a fair approximation of costs and systemic benefits of barrier mitigation were used to attempt to answer pertinent research questions.

Several lines of inquiry were undertaken by using the customised spatial tools and optimisation models to analyse the three selected river networks. First, optimisation analyses were performed for each system for a suite of budgets. Using priorities identified by the optimisation models, the estimated gains to systemic connectivity were plotted against budget. It was expected that estimated connectivity gains from optimal barrier mitigation would increase as budgets increase, but at a decreasing rate, as in previous studies (O'Hanley & Tomblerin, 2005; O'Hanley, 2011; O'Hanley et al., 2013).

Second, the effects of different methods of quantifying river network size on the overall connectivity gains versus budget relationship were then qualitatively examined. Third, a basic sensitivity analysis was undertaken to understand the importance of accurately assessing culvert permeability. Fourth, the estimated relative impacts to systemic connectivity of culverts and hydropower dams, taken individually and as groups, were investigated. Fifth, a related question of whether culverts taken individually or in combination would out-perform hydropower dams in prioritisations for a suite of budgets in the directed and undirected models. Sixth, optimisation models were applied while considering all barriers equally in terms of cost of mitigation. By controlling for cost in this way, the connectivity effects of the mitigation of individual barriers and sets of barriers were investigated. Whether restoration of connectivity at barriers would be likely to exhibit additive or non-additive cumulative benefits to overall systemic connectivity was also explored.

1.4 Thesis Layout

This thesis is presented in monograph format, with five chapters. Chapter One outlines the research problem, questions and objectives. Chapter Two presents a literature review that provides background and context for the research. Chapter Three describes the research methods used. Chapter Four reports results and Chapter Five the interpretation and synthesis of these results. I provide concluding remarks in Chapter Six.

CHAPTER 2: LITERATURE REVIEW

2.1 Longitudinal Connectivity of Rivers

Connectivity in riverscapes is defined as "exchanges of matter (for example, water, sediment, nutrients), energy (for example, organic detritus), and organisms (movement / migration) across the riverine landscape" (Ward, 1997, p.57). *Longitudinal connectivity,* as defined by Ward (1989), is the interactive pathway of upstream-downstream linkages in a river system. Longitudinal connectivity is the driving force behind the *river continuum*, the gradient of physical parameters, and biological patterns and processes from sources to sinks (Vannote et al., 1980). Disruptions to longitudinal connectivity by dams, culverts, and weirs impact riverine ecosystems in a variety of ways (Welcomme & Marmulla, 2008). Impacts of habitat fragmentation include genetic isolation (Pringle, 1997; Gosset et al., 2006; Morita et al., 2009; Horreo et al., 2011), changes in nutrient cycling and primary productivity (Kroeze et al., 2012), biodiversity reduction (Cumming, 2004; Bailey et al., 2007; Ziva et al., 2012) reductions in accessible spawning and rearing habitat (Morita et al., 2009; Beechie et al., 1994), division and isolation of fish populations (Gehrke et al., 2002; Morita & Yamamoto, 2002; Schick & Lindley 2007; Morita et al., 2009; Esguícero & Arcifa, 2010) and impedance of fish migration (Peter, 1998; Gosset et al., 2006; Fukushima et al., 2007; Laffaille et al., 2009). Globally, the presence of anthropogenic fragmentation of river systems is believed to have contributed to the drastic reduction in freshwater migratory fish abundance and biodiversity (Pringle et al., 2000; Nilsson et al., 2005; Liermann et al., 2012).

2.2 Global Fragmentation of Riverine Systems

There is strong evidence that the state of the world's freshwater ecosystems is dire, with biodiversity believed to be declining faster than any other biome (Sala et al., 2000; Dudgeon et al., 2006). Fragmentation of river systems due to damming and other anthropocentric activities is severe, with large dams (>15m) affecting 172 of the 292 largest rivers in the world (Nilsson et al., 2005). One study estimated that there are over two million dams in the United States alone (National Research Council, 1992). Another concluded that human activities have reduced accessible stream habitat in Maine by 80%

as far back as 1860 (Hall et al., 2011). As much as 84% of riverine habitat on the eastern seaboard and Lake Ontario is entrained by dams (Busch et al., 1998).

Numerous small barriers on lower-order streams pose a cumulative effect that is hard to gauge exactly but widely believed to be severe (Warren & Pardew, 1998; Roni et al., 2002; Wheeler et al., 2005; Poplar-Jeffers et al., 2009). A survey of the literature found that culverts are likely a significant source of habitat fragmentation. One study found that Coho salmon smolt production was reduced by 24-34% by impassable culverts and other barriers and that this effect was greater than the combined effect of hydropower and other forest-management practices in the same basin (Beechie et al., 1994). In British Columbia, Canada, there are an estimated 76,000 culverts and a recent survey of 1100 of them found 58% to have a low likelihood of passing fish (Forest Practices Board, 2009). Hicks and Sullivan (2008) found that 55% of culverts in South-western Nova Scotia surveyed (omitting those found on non-fish-bearing streams) posed a barrier to fish passage.

In response to the realization of the extent of the problem both locally (e.g., Figure 1) and globally, there is a growing number of initiatives being made to mitigate the effect of both major and minor barriers to fish passage. Between 1990 and 2003, annual expenditures on river restoration in the United States alone were estimated to be in excess of 1 billion USD (Bernhardt et al., 2005). The European Water Framework Directive sets river continuity as one of the elements that determines the Ecological Status of European rivers and efforts are underway there to restore habitat connectivity (Mader & Maier, 2008). The Columbia River basin in the United States has also been the focus of much expenditure and activity related to increasing river continuity, with over 7 billion USD spent in the last 30 years to save historically large runs of Pacific salmon (Williams, 2008). In British Columbia, between 2008 and 2011, an estimated 11.8 million CAD was spent solely on prioritising and restoring fish passage impacted by forestry roads (Fish Passage Technical Working Group, 2012).

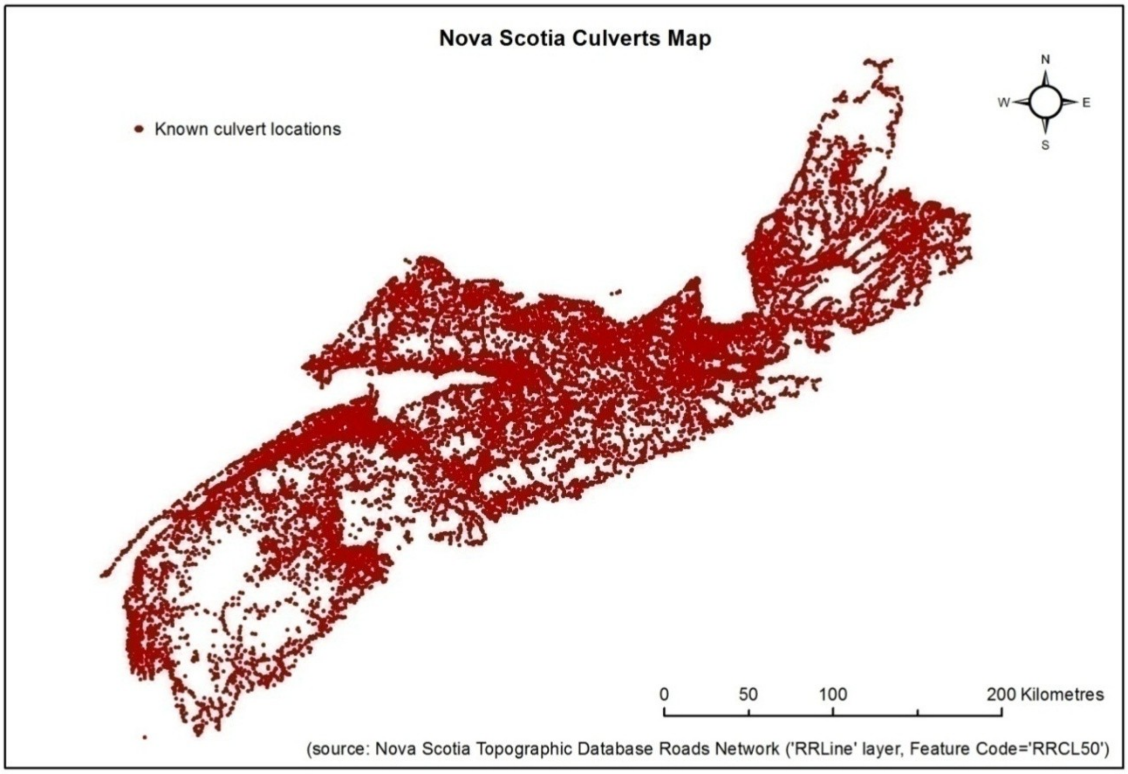

Figure 1: The count of known culvert locations in Nova Scotia exceeds 39,000. Not included are many private and Crown forest-road culverts; actual numbers are likely significantly higher.

2.3 River Restoration Ecology Successes and Failures

In recent years, river restoration ecology has emerged as a distinct field from hydroecology (Palmer & Bernhardt, 2006). The academic fields at the centre of river restoration ecology are engineering, ecology, geomorphology, and hydrology, each with its own conceptual frameworks and problem-solving approaches (Palmer & Bernhardt, 2006). The difficulty of bridging paradigm divides has perhaps contributed to often disappointing results of restoration efforts (see Wohl et al., 2005; Roni et al., 2008). Many blame cases of restoration failure on selection of projects without adequate consideration of the benefit to the larger watershed (Kondolf et al., 2006). A systemic view of restoration is now strongly called for in the literature (see Giller, 2005; Jansson et al., 2007; Lake et al., 2007; Palmer, 2009; Beechie et al.., 2010). However, estimating the watershed-scale effects of restoration is particularly difficult:

> In a similar vein, the sixth frontier is finding new and creative ways to measure the cumulative contribution of individual projects to overall

> watershed improvement. Empirical data and landscape models are needed to prioritise the selection of future restoration sites and to develop basin-scale monitoring approaches that look not at on-site improvements, but catchment-scale changes.
> (Palmer & Bernhardt, 2006, p.3)

Of the techniques applied to river restoration, culvert and barrier removal has shown notable promise at restoring system connectivity and ecosystem processes relative to in-stream habitat restoration, for example (Roni et al., 2008).

2.4 Systematic Conservation Planning: Connections

In an important paper, Roni et al. (2002) proposed a hierarchy of stream restoration, positioning connectivity restoration on the top. However, Roni et al. (2008) amended the hierarchy giving precedence to conservation of intact riverine ecosystems. In the context of river restoration, the priorities therefore are (1) conserve headwaters, (2) restore flow and water quality, (3) restore connectivity, and (4) restore habitat (Roni et al., 2008). It is significant that this framework represents a convergence between restoration ecology and *systematic conservation planning* (Margules & Pressey, 2000; Moilanen et al., 2008; Linke et al., 2011; Turak & Linke, 2011), two subfields that have remained somewhat isolated from one another. In the context of river conservation, systematic conservation planning usually prioritises connectivity upstream to headwaters because the directional flow of the system means, for example, that point-source pollution upstream has disproportionate effects on downstream areas.

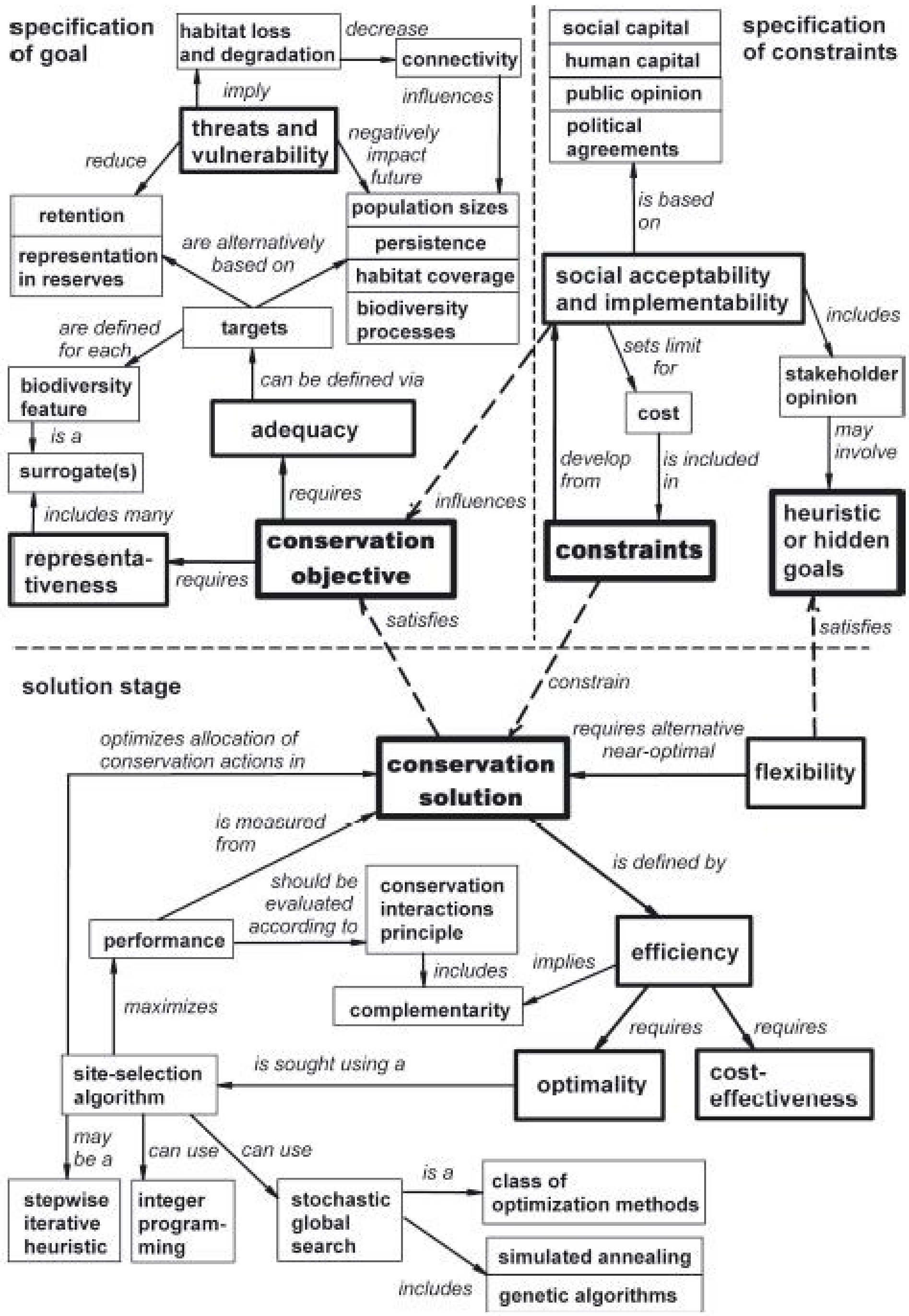

Figure 2: Conceptual framework of systematic conservation planning, created by Moilanen (2008).

Related concepts developed in systematic conservation planning include *spatial efficiency*, *complementarity,* and the *conservation interactions principle (*Moilanen, 2008; Figure 2). Spatial efficiency is a function of complementarity, a component of the conservation interactions principle stating that, "... conservation benefits of all conservation actions across the landscape should be evaluated jointly and account for long-term consequences of interactions between actions" (Moilanen, 2008, p.1657). Reserve-site selection algorithms described in the systematic conservation biology literature employ the conservation interactions principle to account for longitudinal connectivity of river networks (Linke et al., 2007; Moilanen et al., 2008; Nel et al., 2011; Hermoso et al., 2011). For example, Nel *et al.* (2011) accounted explicitly for three types of longitudinal connectivity: "[...] requirements for large migratory species, identification of free-flowing rivers and selection of upstream management zones required to support river reaches selected for achieving representation" (Nel et al., 2011, p.113). The first type of connectivity they described corresponds to anadromous connectivity (what shall here be referred to as *upstream directed)*. The second type identifying free-flowing rivers corresponds to potamodromous connectivity (what shall here be referred to as *undirected*), and the third type corresponds to catadromous (what shall here be referred to as *downstream directed*) connectivity.

The systematic conservation planning literature now widely embraces optimisation as a way to help achieve spatial efficiency in network design (Moilanen et al., 2008; Nel et al., 2008) with its application now emerging in freshwater conservation network planning (e.g., Moilanen et al., 2008; Newbold & Siikamäki, 2009; Nel et al., 2011; Hermoso et al., 2011). In most cases, these applications use software packages with optimisation algorithms embedded. For example, in a systematic prioritisation of freshwater reserves Moilanen et al. (2008) modified the ZONATION software (Watts et al., 2009) that links GIS-based information with species distribution models to account for directed connectivity of river systems.

Restoration ecology has lagged behind systematic conservation planning in the development of algorithms and models that help make the most efficient planning choices:

> "...[A] literature on systematic conservation planning and reserve site selection (RSS) has developed at the interface between ecology, conservation biology, operations research, and environmental economics. These studies use numerical optimisation techniques from

> operations research, often integer programming methods and heuristic algorithms, to prioritise candidate sites for a network of nature reserves to protect species and their habitat."
> (Newbold & Siikamäki, 2009, p.1774)

While the systematic conservation planning community has for some time incorporated complementarity-based algorithms and considered opportunity costs in reserve-site selection (Margules & Pressey, 2000; Rodrigues et al., 2000; Rodrigues & Gaston, 2002; Nel et al., 2008), the freshwater restoration ecology community has not. Additive step-wise site-selection algorithms, similar to the stepwise selection methods currently used in restoration site selection (e.g., Bourne et al., 2011), and step-wise heuristic algorithms were set aside in conservation planning in the 1990's in favour of more robust optimisation and modeling approaches. However, they are still widely used today in river restoration planning.

2.5 Quantifying Longitudinal Connectivity

It is only recently that metrics have been developed to quantify longitudinal connectivity (e.g., Cote et al., 2009). When planning restoration from a system scale, it is important to quantify the benefits to longitudinal connectivity a given project or set of projects will yield (Kondolf et al., 2006; Beechie et al.., 2010). Yet, in prioritisation of barrier removal or mitigation, systemic longitudinal connectivity impacts have most often not been accounted for (e.g., Nunn & Cowx, 2012), accounted for only in a localized way (Taylor & Love, 2003; Kuby et al., 2005; Kocovsky et al., 2009; Zheng et al., 2009), or derived using species richness, abundance, or assemblages as indicators of connectivity (e.g., Mader & Maier, 2008). Review of relevant literature revealed only a few studies that accounted for longitudinal connectivity, though it may be that this is implicitly or explicitly the goal of most restoration, even in the absence of measuring it. O'Hanley and Tomberlin (2005) used *passability-weighted network gains* to maximise the systemic longitudinal connectivity to the ocean (focusing on diadromous fish movement) and O'Hanley (2011) used subnetwork gains to maximise the single largest subnetwork (using binary passabilities; focusing on potamodromous fish movement). Bourne et al. (2011) used the DCI to rank removal for both potamodromous and diadromous longitudinal connectivity. The continuity index was also developed for assessing longitudinal connectivity (Pini Prato, 2007).

One method of measuring longitudinal connectivity in rivers is to assign coefficients of connectivity between river segments (e.g., Moilanen et al., 2008) or to barriers dividing segments of the network (e.g., O'Hanley, 2011). Cases have been described in the literature which use both binary coefficients (Zheng et al., 2009, O'Hanley, 2011) and quantitative, or continuous, coefficients (e.g., O'Hanley & Tomberlin, 2005). As Hermoso et al. (2011) noted, it is more reflective of the actual state of the river continuum (sensu Vannote et al., 1980) to use continuous coefficients of connectivity. Models that use continuous coefficients would also be more amenable to cases where probabilities of connectivity are assigned or are weighted combinations of a number of connectivity coefficients (e.g., between species or directions of travel).

The DCI was developed in response to the paucity of metrics available to quantify longitudinal connectivity (Cote et al., 2009). This metric is important because it can be used to help quantify connectivity benefits at the watershed scale and allows for comparisons of overall connectivity (and overall connectivity gains) between two or more watersheds. It is framed and presented in terms of probabilistic movement, likely stemming from probabilities of capture-recapture applied to many models in ecology and biology. The general DCI formula is:

$$\boldsymbol{DCI} = \sum_{\boldsymbol{i=1}}^{\boldsymbol{n}} \sum_{\boldsymbol{j=1}}^{\boldsymbol{n}} \boldsymbol{c_{ij}}\, \boldsymbol{P}(\boldsymbol{C} = \boldsymbol{c_{ij}}) \qquad (2.1)$$

Where *i* and *j* denote two river segments with c being the coefficient of connectivity between them. This coefficient is the probability of bidirectional passage between the two segments or 'coincidence probability' (sensu Pascual-Hortal & Saura, 2006). The second half of the equation, $P(C=c_{ij})$, reads *the probability that any randomly selected segment pair or path is segment* c_{ij}.

There are two subtypes of DCI, the Potamodromous DCI (DCI_p) and the Diadromous DCI (DCI_d) that correspond to two types of longitudinal connectivity. Put succinctly, the DCI_d accounts for directional movement up or down a river network, whereas the DCI_p accounts for movement within the network regardless of flow direction. The equations for the DCI_p and DCI_d (Cote et al., 2009) are:

$$\boldsymbol{DCI_d = \sum_{i=1}^{n} \frac{l_i}{L}\left(\prod_{m=1}^{M} p_m^u\, p_m^d\right) * 100} \tag{2.2}$$

$$\boldsymbol{DCI_p = \sum_{i=1}^{n} \sum_{j=i}^{n} c_{ij} \frac{l_i}{L}\frac{l_j}{L} * 100} \tag{2.3}$$

The first half of DCI_d equation takes the length *l* of each segment of river *i*, for all segments *n*, scaled to the total length of all segments in the system l_i/L. Alternatively expressed, it is the probability that a randomly placed point in the network will fall within segment l_i The second half of the equation ($\prod_{m=1}^{M} p_m^u\, p_m^d$) takes the set of barriers *M* between each segment *i* and the river mouth, and calculates the product of their permeabilities *p*. The product of the permeabilities is taken between river mouth and any segment *i* because each barrier encountered reduces the chances of passage to the next. For example, the probability of passage between a segment pair with two barriers between, *a* and *b*, with the permeability of both being 0.5 would therefore be 0.25. The permeability of any given barrier is calculated as the product of the upstream and downstream permeabilities ($p_m^u p_m^d$). The product of the upstream and downstream permeabilities is taken because the model is framed in terms of bidirectional movement, that is, the probability that a fish can move from one segment to the second segment and back. Alternatively, it can be conceived as: "the probability that two animals randomly placed within the habitat are able to find each other given the set of habitat patches and links" (Pascual-Hortal & Saura, 2006, p.962).

The DCI_p equation can be read as the sum of all segment pair connectivities (c_{ij} 's) with each scaled by the probability "of observing a particular c_{ij}" (Cote et al., 2009, p. 104).

That is, each segment as a fraction of the network ($\frac{l_i}{L}$) also represents the probability that that segment will be randomly chosen. The probability that a given segment pair is selected randomly is thus the product of the individual selection probabilities ($\frac{l_i}{L}\frac{l_j}{L}$). It is worth noting that one can frame this same problem in terms other than probabilities - measuring connectivity for diadromous and potamodromous movement in terms of absolute habitat. For diadromous movement, the amount of connectivity can be conceived as the *amount of passability-weighted habitat available from the ocean.* This is what O'Hanley and Tomberlin (2005) set out to maximise in the objective function of their optimisation model for diadromous connectivity:

$$\max z = \sum_{j \in J} v_j \, \alpha_j \tag{2.4}$$

In this equation, O'Hanley & Tomberlin (2005) are maximising the sum of habitat accessible upstream of all barriers z in the set of barriers J with each habitat amount v_j multiplied by permeability of all barriers downstream α_j. The method for calculating the total permeability of all barriers downstream is quite similar to that employed by Cote et al. (2009). The O'Hanley and Tomberlin (2005) equation for permeability gains upstream of a barrier:

$$\alpha_j = \prod_{k \in D_j} (\bar{p}_k + \sum_{i \in A_k} p_{ik} x_{ik}) - \prod_{k \in D_j} \bar{p}_k \quad \forall j \in J \tag{2.5}$$

This equation describes the gains in permeability of a given set of projects A_k upstream of a given river segment j. It includes a binary decision variable x_{ik} which is 0 if a project is not chosen and 1 if it is. The benefits of repair in terms of permeability p_{ik} gains are thus measured, rather than just the resulting permeability, but it is essentially the same approach as that of Cote et al. (2009); the permeability to any stream segment α_j from the ocean or a given point in the network is the product of the permeabilities $\bar{p}$ in the set of intermediate barriers D_j . To simplify, when measuring connectivity to the ocean the total permeability is the product of all barriers downstream of segment j:

$$\prod_{k \in D_j} \bar{p}_k \tag{2.6}$$

A notable difference between the approaches is that habitat (i.e. segment length v_j) is not scaled as a fraction of total watershed habitat in O'Hanley and Tomberlin's (2005) approach, but this could easily be done by dividing the accessible habitat v_j by the total habitat is the system. If this were done, and downstream passability were accounted for, then the result would be the $\mathrm{DCI_d}$ for the system. Thus, the optimisation method used by O'Hanley and Tomberlin (2005) can also be used to maximise the $\mathrm{DCI_d}$ (Cote et al., 2009).

In a similar fashion as described above, maximisation of the single largest undirected sub-network (i.e., potamodromous movement; O'Hanley, 2011) happens to also maximise the $\mathrm{DCI_p}$ (Cote et al., 2009). This can be demonstrated on a simple hypothetical network, where all permeabilities are binary, all costs of repair are identical, and the $\mathrm{DCI_p}$ is therefore:

$$DCI_p = \sum_{i=1}^{n} {l_i^2}/{L^2} \tag{2.7}$$

Where l is the length of a given river subnetwork and L is the total network length for all segments (n). Maximising the sum of all l^2/L^2 will result in a single large subnetwork rather than many evenly sized subnetworks (Figure 3). This is due to the squaring of each l and L.

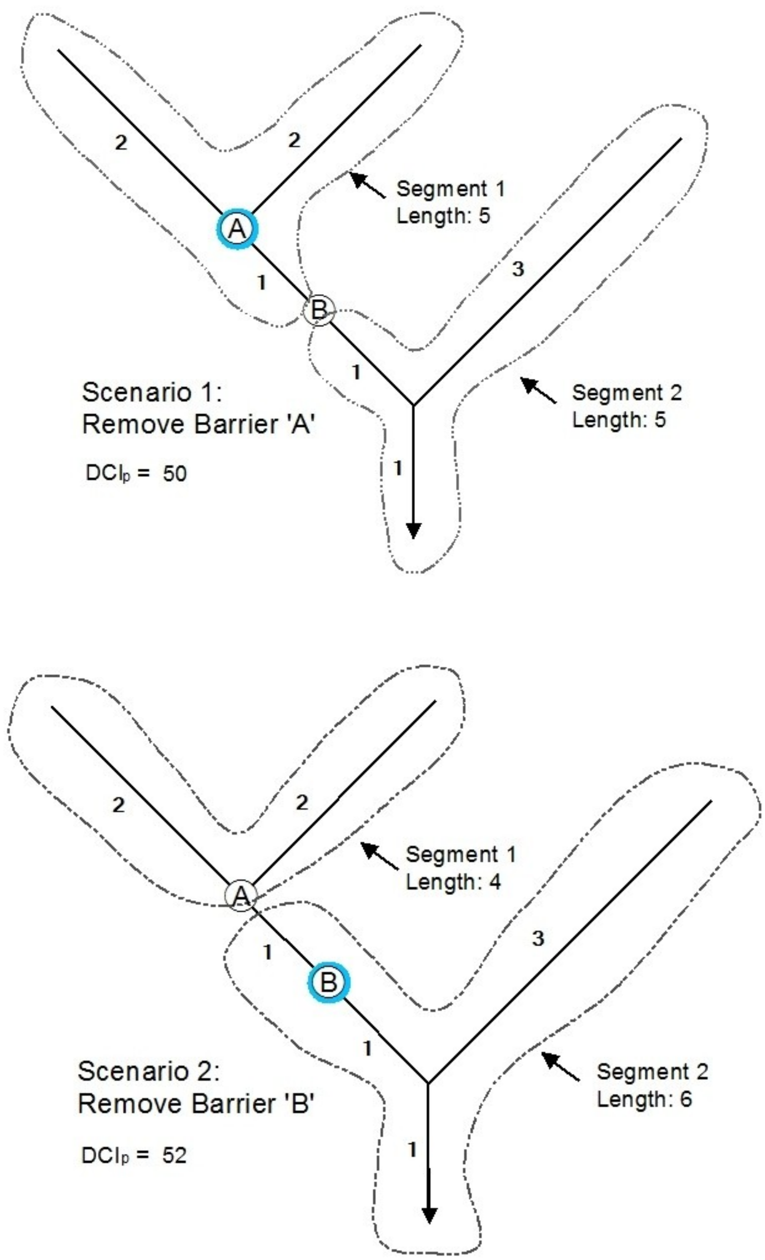

Figure 3: Two scenarios of barrier removal on a hypothetical network, a simple though general illustration of maximisation of DCI_p also tending to maximise the single largest undirected subnetwork. Costs of barrier removal between A and B are considered equal, as are permeabilities. Scenario one depicts a prioritisation resulting in two equally large subnetworks whereas scenario two shows prioritisation for the single largest subnetwork. DCI_p is maximised with creation of largest single undirected sub-network.

Several subtypes of longitudinal connectivity relevant to ecosystem pattern and process are emerging in the literature: the connectivity between downstream and upstream reaches (and conversely the connectivity between upstream and downstream), and the connectivity within the system regardless of direction. The DCI_d, with coefficients of passage for upstream merged with those for downstream (e.g., Cote et al., 2009; Anderson et al., 2012), is a generalised measure of the first two types of connectivity, as is the objective equation used by O'Hanley & Tomberlin (2005). Similarly, the DCI_p is a measure of connectivity within the network disregarding direction (similar to the objective function in O'Hanley, 2011). The terms corresponding to the particular migratory life strategies of fish are often used (i.e., *diadromous connectivity* and *potamodromous connectivity*, respectively (Cote et al., 2009; Perkin & Gido, 2012) though not always (see Nel *et al.,* 2011). The diadromous (i.e., *directed*), subtype could be further divided to anadromous (i.e., *upstream directed*) and catadromous (i.e., *downstream directed*) connectivity, corresponding to the direction of diadromous movement at the time of spawning. For example, salmon are anadromous species and migrate upstream to spawn as adults. In contrast, the American eel is catadromous, and at maturity moves downstream to the ocean to spawn. The important factor for prioritisation is to adjust upstream and downstream permeability values based on the life strategy of the fish being considered.

Differentiation between the two different types of longitudinal connectivity is done implicitly in the development of separate indices of connectivity, but it has not been discussed in a framework beyond fish passage. Indeed, the degree to which decisions overlap between prioritisations of one type of connectivity and another has not been explored in the literature. It is also of particular interest that a primary connectivity objective in *conservation* planning for riverine systems is to maximise connectivity between conserved areas and headwaters (i.e., connectivity between a given reach and upstream reaches; e.g., Moilanen et al., 2008; Newbold & Siikamäki, 2009; Nel et al., 2011; Hermoso et al., 2011). It is not yet clear whether the goal of maximising connectivity to the ocean (common to restoration planning) is a competing or complimentary objective to the conservation planning goal of connectivity to headwaters.

2.6 Prioritising Removal of Barriers

Prioritisation is one of the three major themes of river restoration (Nilsson et al., 2007).

Numerous recent initiatives were found in the literature that prioritised the removal or mitigation of barriers to fish migration. Three methods of prioritisation were noted: (1) scoring and ranking, (2) stepwise SR, and (3) combinatorial optimisation (Table 1). It should be noted that most studies did not explicitly declare methods as falling into one of these three broad categories. Scoring and ranking entails assigning each restoration option a score based on the costs and benefits associated with it and creating an ordered list of projects. Scoring and ranking has the advantages of being relatively quick, flexible, transparent, and does not require a high degree of mathematical or technical expertise, nor specialised software. This method is static, however, in that once a single list of priorities is created, that list is 'locked'. The stepwise variation of SR creates an ordered list, selects projects (usually the single top priority), does that project (or models benefits of doing that project), and creates a new prioritised list. In the stepwise method, upon each iteration the benefits and costs are re-assessed, accounting for project interdependence. Yet the stepwise method does not examine all combinations of barriers. Because optimisation is inherently combinatorial, it is well-suited for assessing projects with a high degree of spatial interdependence. There is evidence that optimisation approaches can yield 25-100% better results than scoring and ranking (O'Hanley & Tomberlin, 2005). In certain circumstances, especially those with a few barriers in quick sequence along the same stretch of network, a divergence in relative efficiency between the three methods quickly occurs as budgets increase (Figure 4). A recent study found conflicting evidence, however, that the interactive (i.e., synergistic or antagonistic) effects were negligible in a simulated river network (Padgham & Webb, 2010), though this conclusion only holds true if no barriers have zero permeability and if movement is volitional.

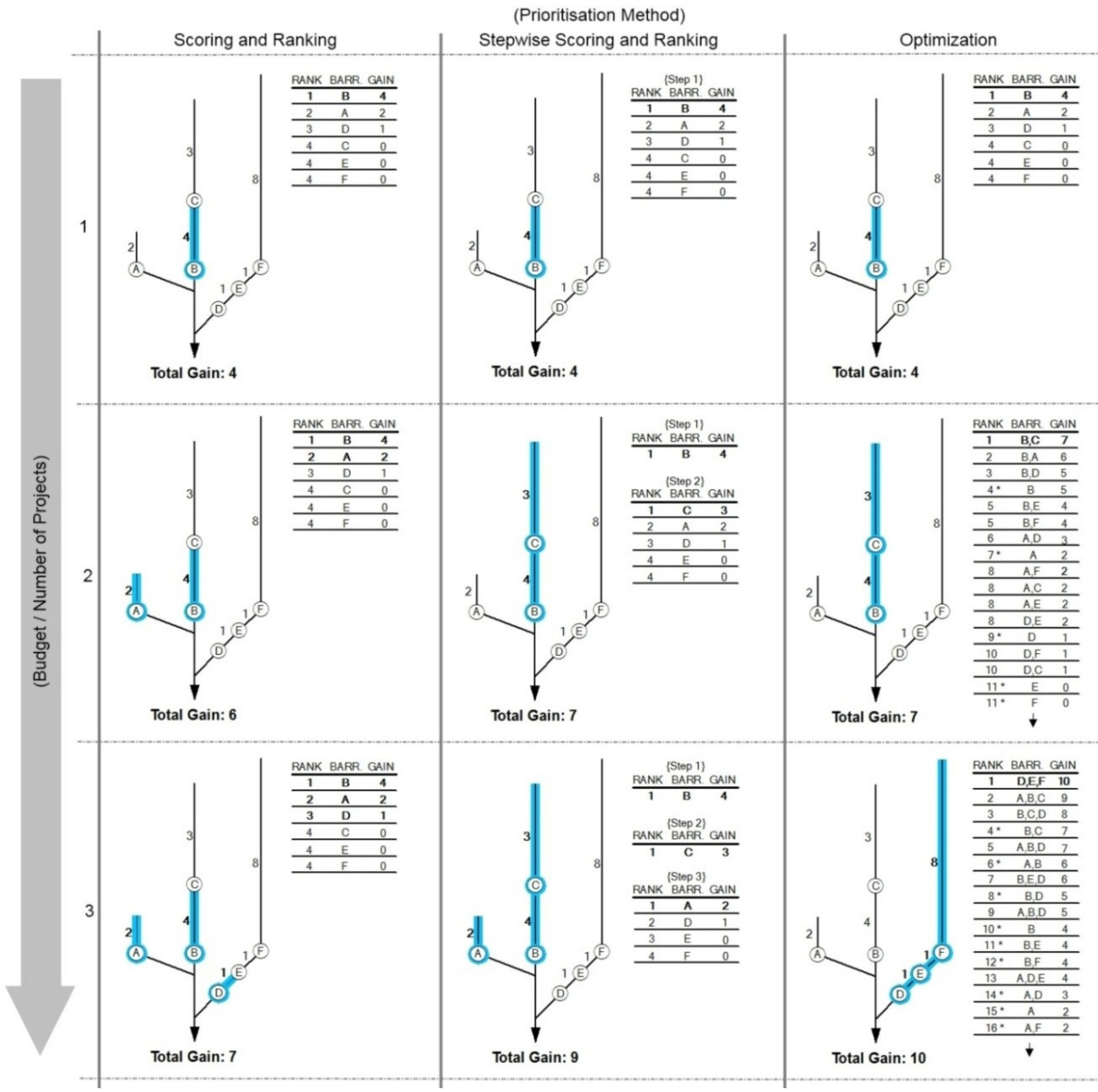

Figure 4: Divergence in efficiency of project selection quickly occurs between three methods of prioritisation. All barriers are assumed equivalent in terms of permeability and cost of removal/repair.

Two local studies of note were found that attempted to evaluate the impact of culverts on fish passage and prioritise their removal in Nova Scotia. In the first study, 60 culverts in the Annapolis River watershed were assessed (Hicks & Sullivan, 2008). The criteria used to assess impact were: whether fish were observed present or absent; the scored habitat quality; whether the barrier fully, partially, or completely blocked passage; and the length of habitat upstream of the barrier until the next. Of the 60 assessed, 37% were found to be full barriers and 18% were partial barriers to fish passage (Hicks & Sullivan, 2008). These results are similar to those found by Langill and Zamora (2002) in a study in

Colchester, Cumberland, Halifax, and Hants counties of Nova Scotia. They assessed 50 culverts on the extent to which they blocked fish passage, assuming that any culverts with a slope greater than 0.5% or perched at their outflow were full barriers to fish passage, and found that 50% were full barriers (Langill & Zamora, 2002). Only the first study by Hicks & Sullivan (2008) attempted to prioritise the mitigation of culverts and used a typical SR technique which, in terms of connectivity, accounted for immediately upstream barriers.

Many studies combine complex habitat and population models into their prioritisation methods. For example, Kocovsky *et al.* (2009) attempted to improve passage for diadromous fish by incorporating segment-specific HSIs, landscape-scale HSIs, length of stream reconnected, and distance from the mouth of the river. They did not incorporate permeability indices, assuming each barrier to be as much of an obstacle as the next. After combining the HSIs, reconnected length, and length to river mouth into a single index, the authors used a SR method of prioritisation (Kocovsky et al., 2009). The only specific weakness they note is that the habitat surveys to create HSIs did not always assess each stream segment between dams (Kocovsky et al., 2009).

2.7 Spatial Decision Support Systems

Decision analyses requiring both professional judgment and quantitative models can be referred to as *semi-structured decision-making problems*, and can be aided by decision support systems (DSS). Decision support systems are designed to help decision-makers solve problems by making sense of data, developing procedures, and modeling the problem along with its constraints. Within a DSS, deterministic optimisation models can be incorporated; however, to solve spatial problems such as the ones presented here, the network structure, directional connectivity, and results must be represented with a spatial component. A spatial decision support system (SDSS) is "…an interactive, computer-based system designed to support a user or group of users in achieving a higher effectiveness of decision making while solving a semi-structured spatial decision problem" (Malczewski, 1999, p.281). An SDSS is comprised of the following components: (1) geospatial data in a GIS, (2) models in a model management system, and

(3) a user interface in the form of a dialogue management system (Figure 5). In the SDSS framework, optimisation models are members of the second component. (Malczewski 1999)

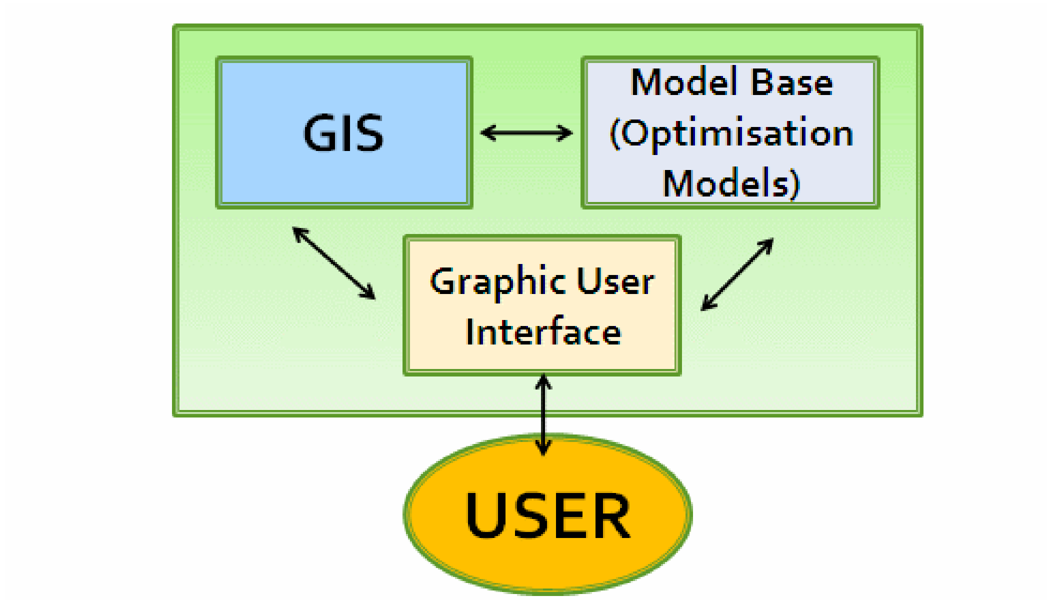

Figure 5: A Spatial Decision Support System (adapted from Malczewski, 1999)

2.8 GIS and Dendritic Ecological Networks

> It is envisaged GIS will be used to facilitate the development of later prioritisation methodologies by enabling assessments of cumulative effects of barriers within stream networks and for comparisons between catchments.
> (Kemp & O'Hanley 2010, p.310)

It is only relatively recently in the study of freshwater landscape ecology that the particular network characteristics of rivers have been incorporated into analytical frameworks and models towards furthering knowledge of the patterns and processes of these systems. In landscape and spatial ecology, network theory has long been applied to understand connectivity between habitat patches (Schumaker, 1996; Bender et al., 1998; Ferreras, 2001; Urban & Keitt, 2001; Moilanen & Nieminen, 2002) but the methods and models applied to terrestrial landscapes are not always suited to analysis of freshwater landscapes (Fagan, 2002; Moilanen et al., 2008). In recent years, the importance of aligning network theory with ecological processes has been emphasized (see Proulx et al., 2005; Peterson et al., 2013). In freshwater landscape ecology, river systems are now conceptualized as dendritic ecological networks (Fagan, 2002; Grant et al., 2007; Peterson et al., 2013) with associated statistics and analysis methods developed specifically for them (e.g., Cote et al., 2009).

Two logical network characteristics (also referred to as *network topologies*) of river systems set them apart from other graph networks: (1) tree-like (i.e., hierarchical) structure and (2) strongly directional flow from 'branches' to the 'trunk' (i.e., *directed graphs*; see Proulx et al., 2005). This leads to a one-to-many relationship between the sources and sink of a given network; that is, one sink (i.e., ocean outflow) has many sources (i.e. headwaters). In addition, two important ecological characteristics of freshwater networks are that (1) branches and nodes can serve as habitat and (2) the network structure itself dictates community assemblages, abundance, and interactions to a much larger degree (Grant et al., 2007). There are few software models or spatial tools available for analysis of river systems that account for these characteristics.

Both the DCI (Cote et al., 2009) and the optimisation model presented by (O'Hanley & Tomberlin, 2005) adopt the DEN concept that observes river systems as a special case of network / graph that has a tree-like structure (Grant et al., 2007). The DEN conceptual framework of rivers combined with the strong directional flow of these systems leads to two noteworthy generalisations: (1) that there is only one path between any two points of a DEN (see Erős et al., 2012), and (2) that any given barrier on a DEN has a maximum of one downstream connected barrier but possibly many upstream connected barriers. These two generalisations lead to advantages in development of algorithms and network models that represent river systems. They also can lead to over-simplifications of actual river network topology, like braided sections of a river (see Ward, 1997). Grant et al. (1997) made note of this, and identified the value of a framework that could have a DEN exhibit a hierarchical pattern of dendritic structure, with occasional sub-patterns of lattice network structure.

Geographic information systems software and toolsets provide useful coupling of visualization, spatial data, and analysis. It has been used extensively in the systematic planning of freshwater conservation areas (e.g., Moilanen et al., 2009; Watts et al., 2009; Pressey et al., 2009) but has only been applied to a limited extent to the problem of barrier removal prioritisation (CalFish, 2011; US Fish and Wildlife Service, 2012). Kemp & O'Hanley (2010) note that although current GIS applications are able to model one or a few barrier removal scenarios in detail, they are ill-suited for system-scale planning with large numbers of barriers. There are two software extensions for the ArcGIS Desktop suite (Environmental Systems Research Institute [ESRI], 2012a) available: RiVEx (Hornby, 2012) and FIPEX (Fisheries and Oceans Canada, 2010). With RiVEx it is possible to analyze the upstream and downstream habitat associated with all barriers in the network, though scenario modeling is difficult because each time a barrier is added or removed from the network, the network must be rebuilt. The Fish Passage Extension uses an in-built feature of ArcGIS, the *geometric network model*, that is designed for analyses of electrical and water distribution, as well as river networks (ESRI, 2012b). The geometric network model provides the ability to turn barriers 'on' or 'off' without rebuilding connectivity in the network (ESRI, 2012b). The Fish Passage Extension (Fisheries and Oceans Canada, 2010) extends upon the geometric network model and provides the ability to include polygons in analyses and exclude river line segments that run through these polygons. However, prior to the research presented in this study, both softwares lack the ability to conduct recursive barrier removal simulations and report gains in terms of overall system connectivity.

2.9 Optimisation in River Restoration

Based on the current literature, it is still unclear exactly where optimisation fits in the framework of decision analysis for river restoration planning. In their categorization of general approaches to restoration prioritisation, Beechie et al. (2008) classified SR and "more complex models" (p.894) together as part of *decision-support systems*. In their extensive review of barrier passage estimation and removal prioritisation literature, Kemp and O'Hanley (2010) separated prioritisation methods into three categories: SR, GIS, and optimisation; however, GIS can be used as a tool within which SR and optimisation prioritisations may be conducted, as Aerts et al. (2003) demonstrated by coupling GIS and integer optimisation models into a simple spatial decision-support system for multi-site land-use allocation. Indeed, Kemp and O'Hanley (2010) acknowledged the potential for combining the two: "The use of optimisation models combined with GIS shows enormous promise for supporting barrier prioritisation in an effective and cost-efficient manner" (p. 318).

There are various possible reasons for optimisation remaining under-employed in river restoration despite the demonstrated benefits, including:

1. high computational burden,
2. inflexibility of existing models to different scenarios of data availability,
3. a lack of transparency to decision makers

(O'Hanley & Tomberlin, 2005; Beechie et al.,2008)

2.9.1 High Computational Burden

Looking to the systematic conservation planning literature, at the time optimisation approaches were introduced (circa 1996), the computing power required to solve realistically sized problems meant that such problems were often intractable and so heuristic approaches were often heavily relied upon (see Pressey et al., 1996; Pressey et al., 1997). In just six years, by 2002, computing power had increased enough that realistic-sized problems of conservation network selection could then be solved to optimality (Rodrigues & Gaston, 2002). However, at present, there are still hard limits to the size of problems. O'Hanley and Tomberlin (2005) demonstrated the combination of optimisation with heuristic algorithms, which can reduce solve-time considerably. The limit to the size of the *Fish Passage Barrier Removal Problem* (sensu O'Hanley & Tomberlin, 2005) that can be solved to optimality or near-optimality is unclear and depends on the approach taken.

2.9.2 Inflexibility of Existing Models

A review of the literature revealed only two optimisation models for maximising directed longitudinal connectivity (Kuby et al., 2005; O'Hanley & Tomberlin, 2005) and two for undirected longitudinal connectivity (O'Hanley, 2011; O'Hanley, 2013). The directional model created by O'Hanley and Tomberlin (2005) is formulated non-linearly, requiring more advanced algorithms and solvers. The authors developed and utilised a customized dynamic programming optimisation model and, separately, integrating a local search heuristic score and rank procedure (O'Hanley & Tomberlin, 2005). The downside of this approach, as note by O'Hanley and Tomberlin (2005), is that these methods are not easily applied without specialised mathematical and programming expertise. A simpler approach was taken by Kuby et al. (2005) who formulated the problem as a linear model. Their model included the constraint that no dam be removed if a downstream dam is present. Kuby et al. (2005) also restricted permeabilities to binary values, making it insensitive to scenarios where continuous permeabilities can be estimated. The model for maximising the single largest undirected sub-network described by O'Hanley (2011) also used only binary permeabilities. Many restoration prioritisation projects invest a significant amount of resources to estimating passage efficiency and potential gains of various mitigation measures, for which specialized software has been developed (e.g., US Forest Service, 2003). In these cases, models that can incorporate continuous permeabilities are desirable. Lastly, O'Hanley et al. (2013) incorporate continuous permeabilities in a nonlinear program for maximising undirected connectivity; however, as an addendum they adapt the model to a linear program.

When seeking to create a flexible and generic optimisation model, the linear programming method is often preferred, as it is "... the most natural mechanism for formulating a vast array of problems with modest effort... [Linearity] is the only functional form that will be equally applicable (or non-applicable) in a class of similar problems" (Luenberger, 2003, pp.2-3). Integer linear programming (ILP) is a type of linear programming (also known as linear optimisation) technique where the variables are required to be integer only. When some variables are allowed to be non-integer and others are restricted, this is referred to as *mixed integer linear programming* (MILP). There is a rich history behind LP, ILP, and MILP dating back to World War II (see Murthy, 2005) and a diversity of algorithms and software available to solve such problems. The ILP approach has been successfully used in reserve selection and Rodrigues and Gaston (2002) make note that there is "... great flexibility in the type of data and concerns that can be integrated in linear integer problems, while retaining the accountability of the decision process" (p. 128). This stands in contrast with the nonlinear technique which at present cannot be easily solved without expensive software and mathematical expertise, as acknowledged by O'Hanley and Tomberlin (2005)**.**

In addition to inflexibility to data robustness, existing models are not presented within any software package or decision-support toolset. In the systematic conservation planning context, software such as ZONATION (Moilanen et al., 2009), MARXAN (Watts et al., 2009) and C-Plan (Pressey et al., 2009) are available and allow those without a specialisation in programming or operations research to perform analyses. They also provide a framework for data standardization. This is currently lacking in the systematic river restoration planning context. One large problem of riverine analysis is generating network topology for use in model analyses. Currently, few tools exist to analyze river networks quickly and describe the node-edge topology required for robust modeling.

2.9.3 Transparency to Decision-Makers

One remark by Beechie et al. (2008) was that simple prioritisation approaches are more advantageous because they are relatively transparent to decision-makers. Optimisation and more complex models may indeed yield mysterious results because they process 'behind closed doors'. There may be some cluttering of concepts here of *prioritisation* with *scoring*, though, as complex methods for assessing benefits and costs of decisions are a separate thing from the algorithm used to find the best combination of costs and benefits. There is certainly nothing keeping simple or complex simulation scoring and simulation models from being incorporated in the decision analysis of an optimisation model; as Wurbs and Yerramreddy (1994) noted, these two approaches are often combined when applied to river-basin decision analyses. It may be, however, that complex simulation models (e.g., Zheng et al., 2009) combined with optimisation lead end-users to have difficulty understanding the decisions reached. Overall, there appears to be some confusion in the literature and with practitioners surrounding the separation of scoring methods (i.e., assessing benefits or costs of restoration) and prioritisation methods (e.g., ranking, stepwise ranking, optimisation, etc.). As illustration, Beechie et al. (2008) described a four-step process to identifying and prioritising restoration goals: (1) setting a clear goal for restoration, (2) choosing a prioritisation scheme, (3) using watershed analysis to identify restoration actions, and (4) prioritising actions based on assessment. In this framework, the second step refers to scoring the benefits and costs of each restoration project, with the prioritisation implied to be a simple ranking procedure. In the fourth step, "[i]deally, the prioritisation of restoration actions simply involves following through on the approach chosen in step 2, based upon the information collected in step 3" (Beechie et al., 2008, p.897). In conclusion, it is important that optimisation models be presented in this context simply as a combinatorial method of assessing alternatives given a budget and separate from methods to estimate benefits or costs.

2.10 Estimating Permeability

The terms used to describe the degree to which discrete barriers affect longitudinal connectivity vary. The majority of studies measure longitudinal connectivity impacts with a bias towards diadromous and economically significant fish (Kemp & O'Hanley, 2010), although barriers also affect transport of nutrients and sediment (Kroeze et al., 2012). *Attraction efficiency* and *passage efficiency* are terms often used when studying the effectiveness of fishways (Noonan et al., 2011; Smith & Hightower, 2012; Thiem et al., 2012) and refer to the proportion of fish entering and exiting a fishway, respectively. Barrier *passability (*e.g., O'Hanley & Tomberlin, 2005) and *permeability* (e.g., Cote et al., 2009) are also used to refer generally to the degree of blockage a barrier imposes. Cote et al. (2009) integrated permeability in a probabilistic modeling framework that estimates the chances a fish can move freely throughout the network. In this model, degree of passage is conceptualized as bidirectional; the probability a fish can move past a barrier and then back, with permeability of a barrier therefore being the product of the upstream and downstream permeability (Cote et al., 2009). Keefer et al. (2009) derived *passage probabilities* of Pacific lamprey from tracking data. For the purposes of this study, the term *permeability* will be used to refer to the degree to which barriers impact longitudinal connectivity. This choice was made because *permeability* somewhat better encompasses connectivity impacts beyond fish passage such as sediment and nutrient transport.

The choice of criteria and methods used to assess culvert permeability is highly variable in the literature. Anderson et al. (2012) showed that out of 256 culverts surveyed, the percentage of culverts found passable to small-bodied fish varied between 35% and 78%

depending on the method used. Methods differ depending on the situation and range from simply accounting for their presence to robust modeling for individual species. In many cases, barriers are so numerous that it is not practical to do an onsite assessment of each. Additionally, permeabilities are not easily predictable through modeling without field surveys. It is thus often the case that all barriers are treated as either a potential barrier or not. For example, in a prioritisation of >300,000 culverts, Mount et al. (2011) considered all culverts as equally likely to impair passage and eliminated barriers from consideration if species of concern were not likely to be present upstream.

In studies where it is possible to visit all barriers, scores are often assigned (e.g., Mader & Maier, 2008; Hicks & Sullivan, 2008; Solà et al., 2011; Nunn & Cowx, 2012) such as '1 - passable', '0.5 - partially passable', '0 - impassable.' These approaches are often based on heuristics or expert opinion. In some instances, in-depth modeling is done to estimate continuous passage probabilities, taking into account variables such as physiology and fitness among individuals and between species, hydrology and flow dynamics, and physical attributes of the barrier (see Kemp & O'Hanley 2010 for an overview). An alternative method is to derive permeability from presence-absence and tracking surveys (Bourne et al., 2011; Nislow et al., 2011; Perkin & Gido, 2012; Pépino et al., 2012). Robust methods of estimating permeability of barriers may be important; "Since biological communities change gradually through natural longitudinal gradients in rivers, spatial connectivity is better addressed through continuous probabilities..." (Hermoso et al., 2011, p.66). A given barrier could therefore have many permeability values depending on the data available and objectives of the decision-making scenario. For example, factors contributing to permeability that may be taken into account may include fish species (Peake et al., 1997; Porto et al., 1999; A. Haro et al., 2004; Peake, 2008), physical attributes of the barrier (Porto et al., 1999; Vander Pluym et al., 2008; Mueller et al., 2008), and in-stream flow characteristics (Reiser et al., 2006; Rolls, 2011). Permeability also varies directionally (Peake et al., 1997; Thiem et al., 2012), inter-specifically (Porto et al., 1999; Kondratieff & Myrick, 2006; Mueller et al., 2008), intra-specifically or within a given population (Brett, 1971; Holthe et al., 2005), environmentally (Rolls, 2011), and temporally. However, it is most often not possible to

Table 1: Literature Review Summary of Prioritisation Measures

Prioritisation Method	Example Study	Connectivity Measure (s) / Method (s)	Permeability Measure	Number of Barriers	Types of Barriers
Scoring and Ranking	Karle (2005)	downstream barriers present	Continuous parameter (FishXing)	n/a	n/a
	Hicks & Sullivan (2008)	'presence of upstream barrier' score	score (three classes)	268	Culverts
	Mader & Maier (2008)	None	score (3 classes)	230	Weirs, Falls, Debris
	Kocovsky et al. (2009)	distance from river mouth	Binary parameter (assumed 100% impassable)	20	Dams
	Poplar-Jeffers (2009)	None	Continuous parameter (FishXing)	120	Culverts
	Pini Prato et al. (2011)	'Continuity Index' (CI; Pini Prato, 2007))	Score (3 classes)	16	Small obstacles
	Nunn & Cowz (2012)	likelihood of access (downstream barrier passage)	score (5 classes)	67	Weirs
	Anderson (2012)	presence of upstream / downstream barrier	Continuous parameter / probabilistic	156	Culverts
Scoring and Ranking (stepwise)	Taylor & Love (2003)	presence of upstream barrier	score (5 classes)	n/a	Culverts
	Diebel et al. (2010)	Custom Connectivity Status Metrics (C^{inv}, C^{avg})	Continuous parameter	121	Road Crossings
	Mount et al. (2011)	Number of barriers downstream	Binary parameter (assumed 100% impassable)	>300,000	Culverts
	Bourne et al. (2011)	DCI_p&DCI_d	many (experimental)	43	Culverts
Combinatorial Optimisation	Kuby et al. (2005)	Presence of downstream barrier	Binary parameter	150	Dams
	O'Hanley & Tomberlin (2005)	Connectivity Matrix	Continuous parameter	289	Culverts
	Zheng et al. (2009)	Presence of downstream barrier	Binary parameter	139	Dams
	O'Hanley (2011)	Connectivity Matrix	Binary parameter	125	Culverts, Dams (2)

account for all factors contributing to barrier permeability due to lack of data, expertise, resources, or the complexity it adds to prioritisation models, and so binary values are commonly used (see Table 1).

In general, it is not yet possible to predict culvert permeability without site visits. To date, there are few models that exploit correlations between commonly available geospatial data such as road type, gradient, or stream order and permeability (though see Januchowski-Hartley et al., 2013). Instead, most variables used to estimate permeability are highly variable between sites and are likely determined by installation procedures used at the time of construction. The most common attributes collected in preliminary scoring assessments include outflow drop, length, substrate, flow characteristics, gradient, and outflow pool depth (Table 2).

Table 2: Survey of Culvert Attributes Collected during Preliminary on-site Assessments.

Attribute	Studies
outflow drop	Taylor & Love, 2003; Hicks & Sullivan, 2008; Poplar-Jeffers, 2009; FPB, 2009; Anderson et al., 2012
culvert length	Taylor & Love, 2003; Poplar-Jeffers, 2009; FPB, 2009
culvert substrate	Taylor & Love, 2003; Poplar-Jeffers, 2009; FPB, 2009; Bourne et al., 2011
flow characteristics	Hicks & Sullivan, 2008; Poplar-Jeffers, 2009; Anderson et al., 2012
gradient	Taylor & Love, 2003; Poplar-Jeffers, 2009; FPB, 2009; Bourne et al., 2011
outflow pool depth	Hicks & Sullivan, 2008; Bourne et al., 2011

On the whole, permeability assessments are biased towards upstream permeability and salmonid species, often neglecting downstream permeability and other species (Kemp & O'Hanley, 2010). Downstream permeability is important for many species (Calles & Greenberg, 2009) but especially for catadromous species like the American eel, which is in decline (Hodson et al., 1994; Haro et al., 2000; Cairns et al., 2008) with barriers to migration believed to be a contributing factor (Busch et al., 1998; Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2006a; Machut et al., 2007; Cairns et al., 2008)**.** It is therefore important that longitudinal connectivity at barriers be assessed in a way that encompasses resident fish and other essential ecosystem processes (Ward, 1997; Kroeze et al., 2012).

2.11 Estimating Costs

Costs of mitigating barriers are highly variable and dependant on a number of factors. Bernhardt et al. (2005) found the median project cost for fish passage projects in the U.S. was approximately 30,000 USD. However, their study did not distinguish between projects on large dams and those on smaller barriers such as culverts. A common industry method assumes costs of installing common fish passage structures at dams are a function of the height of the barrier, given that it dictates the length and size of structures. Pool and weir or denil fishways are estimated to cost 20,000 - 30,000 USD per foot of dam height (Connecticut River Watershed Council Inc., 2000; Rhode Island Habitat Restoration Portal, 2003). The cost estimated by Nova Scotia Power Incorporated (NSPI) is 20,000 CAD per foot of height (K. Meade, personal communication, March 18, 2010). A common function for estimating cost of installing a pool and weir or denil fishway (and converting from feet to metres) is:

$$y = 3.28x + 20{,}000 \tag{2.8}$$

Where:

y = *total estimated project cost*
x = *dam height.*
(Connecticut River Watershed Council Inc., 2000; Rhode Island Habitat Restoration Portal, 2003; K. Meade, personal communication, March 18, 2010)

A much more robust model was developed by Zheng et al. (2009) who identified relationships between cost of removal and dam height, length, purpose, and construction material, shown in Eqn. (2.9)

$$\ln\left(C_j^{dam}\right) = \underbrace{7.79}_{(\text{S.E.}=0.52)} + \overbrace{\underbrace{0.80}_{(0.14)}\left(\ln\left(Height_j\right)\right) + \underbrace{0.33}_{(0.11)}\left(\ln\left(Length_j\right)\right)}^{\text{DamSize}} + \overbrace{\underbrace{1.49}_{(0.26)}\left(FuncW_j\right)}^{\text{DamPurpose}} - \overbrace{\underbrace{0.44}_{(0.22)}\left(TypeE_j\right)}^{\text{DamType}} \tag{2.9}$$

The cost of culvert mitigation is highly variable and not easy to estimate. One method of cost estimation associated with culvert projects is to assess cost based on length of river restored, with one case reporting culvert restoration projects costs ranging from 1000 CAD/km to 50,000 CAD/km of river gained (Parker, 1999). Parker (1999) used a rough estimate of 10,000 CAD per project. In contrast, a more recent study found installation of high-arch culverts to cost between 28,000 USD and 50,000 USD (Long, 2009). Another recent cost estimate placed the average cost of replacing impassable culverts at an average of 100,000 CAD (Fish Passage Technical Working Group, 2012). Reasons for such varied estimates stem from variable and un-catalogued methods used during culvert installation, which have a high degree of impact on remediation costs.

2.12 Estimating Habitat Quantity

To prioritise barriers for mitigation, it is necessary to quantify river network feature size. How the size of river network is typically conceptualized, estimated, and incorporated into models varies. Network length and area are the two primary spatial measures used. Studies that use network length in prioritising restoration (O'Hanley & Tomberlin, 2005; Hicks & Sullivan, 2008; Mader & Maier, 2008; Kocovsky et al., 2009; Mount et al., 2011; Anderson et al., 2012; Nunn & Cowx, 2012; Fish Passage Technical Working Group, 2012) are far more prevalent than area (Kuby et al., 2005; Zheng et al., 2009). In contrast, reserve-site selection in a freshwater conservation planning context calls for area quantity measures (Moilanen et al., 2008; Newbold & Siikamäki 2009; Hermoso et al., 2011; Nel et al., 2011), since lateral connectivity between terrestrial and freshwater landscapes requires the protection of land to ensure protection of freshwater (Pringle

2001; Pringle 2003). In the literature, only one case could be found where upstream drainage area was used in prioritising barriers for mitigation (Kuby et al., 2005).

Most efforts to re-establish longitudinal connectivity have a bias towards benefits to salmonid species (Kemp & O'Hanley 2010) as they are important both economically and to ecosystem processes such as nutrient cycling (Naiman et al., 2002; Kemp & O'Hanley 2010), and are in decline in many parts of the world (Amiro, Gibson & Drinkwater 2003; Ugedal et al., 2008; Limburg & Waldman 2009). Consequently, as shallow pools and riffles are characteristics of spawning habitat, length of stream habitat made available has thus been the primary measure of habitat quantity, ignoring lake habitat. Additionally, geospatial data on areas of streams are not generally available because they are usually digitized from aerial photos which limits resolution. A cursory examination of data available in the Nova Scotia Hydrographic Network (NSHN; Service Nova Scotia and Municipal Relations, 2012) shows that streams are represented by lines rather than polygons for stream widths less than about 27 m. However, for species which use lotic (i.e., lake) environments or show little preference (e.g., American eel), considering length of habitat reconnected alone is not as representative of habitat for these species as area measures.

2.13 Estimating Habitat Quality

Estimating river and stream habitat quality for fish is a niche subfield of its own. Basic scores are often assigned using environmental data collected for river segments (Hicks & Sullivan, 2008; Nunn & Cowx, 2012). Numerous robust, species-specific models have been developed to estimate relative habitat suitability at various scales (e.g., Amiro, 2006). A common approach is to weight sections of river using an HSI.In a context similar to this study, Kocovsky et al. (2008) develop HSIs using macro-invertebrate food-source abundances they later used to weight mitigation priorities (Kocovsky et al., 2009). Newbold and Siikamaki (2009), in a river reserve-site selection process, used pollution and land use upstream of sites to weight habitat. Some approaches use fish presence-absence data as either a surrogate for habitat quality (Mount et al., 2011; G. Anderson et al., 2012), or to detect key isolated populations to reconnect (Tsuboi et al., 2010).

Abundance per stream length was used by Bourne et al. (2011) to weight connectivity measures. Metrics of ecological integrity or status can also be used (Mader & Maier, 2008; Zheng et al., 2009). In the case of the American eel, freshwater habitat presence appears limited more by access than environmental conditions and dispersal models may be more suited for predicting presence and abundance than HSI models (Smogor et al., 1995). However, studies in the literature demonstrating new prioritisation methods have typically omitted measures of habitat quality, making note, however, that habitat can be weighted using measures of relative habitat quality or representativeness of habitat for key species (O'Hanley & Tomberlin, 2005; Kuby et al., 2005; O'Hanley, 2011).

2.14 Cumulative Effects of Longitudinal Connectivity Restoration of Rivers

> Further, assessment methods previously developed to address ecological degradation might effectively be applied to the reverse situation: evaluating ecosystem restoration.
> (Diefenderfer et al., 2011, p.113)

There is no consensus on a steadfast definition of what constitutes *cumulative effects* (see Duinker et al., 2013, for a review). In particular, there is ambiguity in the relevant literature about whether *cumulative effects*, by definition, encompasses both additive (i.e., simple) and non-additive (i.e., complex) effects of ecological stressors and whether it refers to interactions between more than one type of stressor, as opposed to a single stressor. For example:

> Cumulative impacts (or cumulative effects) are defined for the purpose of this study as "the outcomes of numerous pathways of influence initiated by the interactions between multiple human activities in shared space and time. These outcomes may be positive or negative, additive or interactive and may have social economic or environmental implications."
> (Krzyzanowski, 2011, p.253)

In contrast, the definition given by Houle et al. (2010, p.420; citing Riffell et al., 1996) is relatively narrow: "cumulative effects occur when the joint effects of features in close proximity are greater or less than theinfluence of either of the features alone." Furthermore, as is apparent in the above quote from Krzyzanowski (2011), the terms *effects* and *impacts* are often used interchangeably, though there has been a notable shift towards the replacement of the latter with the former (Diefenderfer et al., 2011). Sheelanere et al. (2013, p. 67) noted that the "current practice of watershed cumulative effects assessment and management is simply not working." Reasons given include the difficulty in accounting for effects at the watershed scale (Noble et al., 2011; Seitz et al., 2011) and, generally, issues with assessing cumulative effects at the regional rather than site or project-specific scale (Duinker & Greig, 2006). To meet this challenge, there are significant and ongoing efforts to incorporate landscape-scale effects into cumulative effects assessment (e.g., Seitz et al., 2013; Squires & Dubé, 2013). The study of systematic restoration of longitudinal connectivity of rivers via barrier mitigation is also, depending on the particular definitions adopted and some methodological caveats I will subsequently visit, a study of the cumulative effects of fragmentation due to anthropogenic barriers. The scope of this study differs from most studies of cumulative effects (though see Diefenderfer et al., 2012) in that it addresses only a single stressor, namely anthropogenic barriers, and focuses solely on the systemic connectivity response of the simulated removal of these barriers.

Thresholds of connectivity response to restoration action are a particularly important component of the study of cumulative effects (see, for example, Schultz, 2010; FPB, 2011; Duinker et al., 2013). Again, the approach taken in this thesis research is notably different with respect to thresholds: thresholds are those of systemic connectivity, responding to restorative effect of effort rather than the degrading effect of the addition of ecological stressors. Nonetheless, the results and the methods proposed for evaluating combined response of restoration efforts have potential to contribute to the ongoing study of cumulative effects.

Of particular relevance to both the study of cumulative effects of river and systematic restoration planning is whether restoration actions yield responses that display synergisms or antagonisms. Folt et al. (1999) provide a guiding definition: synergism is defined as interaction between stressors whereby the combined impact of several stressors is greater than what would be predicted by the sum of each taken individually (antagonism is the opposite). Three categories of interactions between stressors, following Folt et al. (1999), are: additive, synergistic, and antagonistic – the latter two often referred to as non-additive (e.g., Crain et al., 2008; Darling & Cote, 2008) or nonlinear (e.g., Diefenderfer et al., 2012). Non-additive interactions between riverine stressors have been observed (e.g., Townsend et al., 2008) and in a study of lateral connectivity restoration, Diefenderfer et al. (2012) found synergisms between dike breaches that led to greater restoration results than predicted by an additive model (Figure 6). Besides advancing general knowledge, identification of the type of responses stressors elicit has relevance to the choice of prioritisation method; SR and stepwise SR are ill-suited to detect and account for responses that are non-additive.

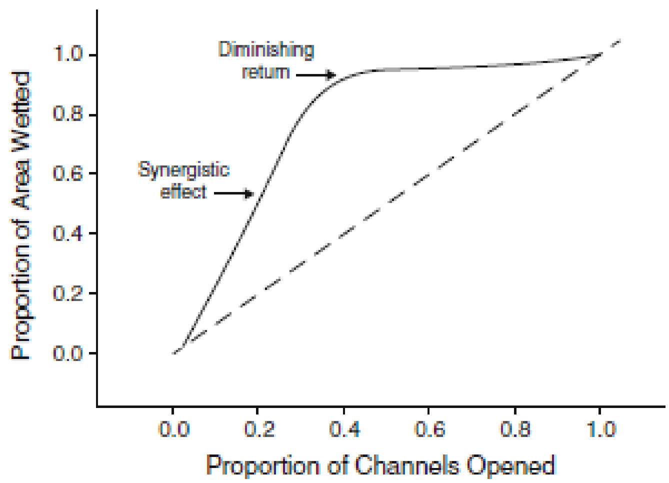

Figure 6: Evidence of synergism followed by diminishing returns of dike breaches in a tidal freshwater tributary of a floodplain of the Columbia River (from Diefenderfer et al., 2012). The dashed line is predicted by a simple relationship.

Optimisation is an inherently combinatorial approach which considers the net total benefits of sets of projects rather than individual projects, and can therefore be used to identify non-additive interactions and cumulative effects. Thus, the results of optimisation compared between two or more budget increments can be used to test a number of relevant hypotheses. For example, an expected observation of optimisation results is that as effort increases the gains in terms of connectivity per unit effort decreases, since optimisation prioritises the best possible combination of projects for any given budget. If non-additive interactions between restoration projects are present, the addition of barriers to optimal decision sets as budgets increase may yield greater returns in terms of connectivity per dollar of budget spent. Two caveats are that by both (1) incorporating variable costs into analyses and (2) restricting decisions to binary values (projects are either 100% chosen or not done at all), thresholds can occur wherein projects that yield a highly efficient return per unit of budget do not appear in optimal decision sets until they are affordable, resulting in sudden increases in marginal gains between budgets. Thus, observations of increasing connectivity gains per dollar of budget spent in optimisation results may be attributable to either (1) cost effects or (2) non-additive synergies between restoration projects. To my knowledge, all past studies that applied optimisation methods to the problem of longitudinal connectivity restoration have made these two assumptions: that effort is measured in terms of cost and barrier repair is a binary variable (Kuby et al., 2005; O'Hanley & Tomberlin, 2005; Zheng, et al., 2009;O'Hanley, 2011). Therefore, it is actually the cumulative (beneficial) effect of *budget* as applied to the problem of longitudinal connectivity restoration that is reflected in the results of these studies. The relative costs between projects may thus account for any observed patterns of increasing connectivity returns per dollar of budget spent, or *economies of scale* (see Diefenderfer et al., 2012 for discussion and contextualization of this term).

To address the cumulative effects of the mitigation of a single ecological stressor such as riverine barriers and isolate non-additive synergies of restoration efforts, optimisation may be applied with relative project costs omitted; that is, all restoration projects may be assigned equal cost. With the economic factor omitted, restoration decisions may remain binary and barriers continue to represent discrete but variably sized packets of a single type of ecological stressor. Although Diefenderfer et al. (2012) did not employ optimisation – rather, a series of randomised barrier removals as the total barriers removed were incremented – the objective was to detect synergies between dike breach in restoring lateral connectivity, which they did by omitting cost and treating number of barriers removed as the discrete unit of both restoration effort and, conversely, ecological stress.

CHAPTER 3: METHODS

3.1 Study Area

3.1.1 Nova Scotia

Nova Scotia is a province on the Atlantic Coast of Canada. Approximately ten percent (5,300/55,344 km^2) of its surface area is freshwater (derived from the Nova Scotia Topographic Database (NSTDB) on features coded 'lakes', 'river-lakes', 'non-coastal rivers'; GeoNova, 2012). There are 46 major watersheds in the province (Nova Scotia Environment, 2011) draining into the Bay of Fundy, the Atlantic Ocean (South Shore), and the Northumberland Strait. Nova Scotia Power Incorporated owns and operates 33 hydroelectric power generation stations on 17 water systems in the province (Nova Scotia Power Incorporated [NSPI], 2009b). The company maintains an estimated 165 dams, wing dams, and flow-altering structures associated with its operations, with a total drainage area of approximately 7306 km^2 (NSPI, 2009b).

Out of 43 species of fish recorded in the freshwater systems of Nova Scotia, there are known to be eleven species that employ a *diadromous* life strategy (i.e., move between ocean and freshwater habitats). Ten of these are native species that spend the majority of their life at sea and move to freshwater to spawn (i.e., *anadromous* fish):

- Atlantic salmon (*Salmosalar)*
- Striped bass (*Moronesaxatilis*)
- American shad (*Alosasapidissima)*
- Gaspereau (*Alosapseudoharengus)*
- Rainbow/American smelt (*Osmerusmordax)*
- Blueback herring (*Alosaaestivalis*)
- Atlantic sturgeon (*Acipenseroxyrinchus*)
- Sea lamprey (*Petromyzonmarinus*)
- Atlantic tomcod (*Microgadus tomcod*)
- Atlantic whitefish (*Coregonushuntsmani*)
  (Davis & Browne, 1996)

The American eel (*Anguilla rostrata*) spends its adult life in freshwater and migrates to the ocean to spawn (i.e., a *catadromous* fish). At least 15 native fish species which move only within freshwater systems (i.e., *potamodromous* fish) are also present in Nova Scotian watersheds (Davis & Browne, 1996).

Nova Scotia's rivers were historically home to large runs of economically significant fish species, many of which are in decline. The Atlantic salmon populations in the Inner Bay of Fundy (IBoF) region were as high as 40,000 in the mid-1980's and were reduced to fewer than 100 by 2003, a decline of more than 99% (Amiro et al., 2003; Anderson et al., 2000; Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2006b). Causes for the decline are still not fully understood but blockage of spawning habitat by local anthropogenic barriers is believed to be a contributing factor (COSEWIC, 2006b). The IBoF population of Atlantic salmon is now considered Endangered (COSEWIC, 2006b).

The American eel is also in decline, and was designated a Species of Special Concern in April 2006 (COSEWIC, 2006a). The population has declined 99% in the Upper St. Lawrence River and Lake Ontario since the 1970's (COSEWIC, 2006a). There are indications of population decline in Nova Scotia (Prosper & Paulette, 2002; Cairns et al., 2008), although there is a paucity of data - the commercial eel fishery in Nova Scotia, from which abundance estimates are derived, is still relatively young. Natural and anthropogenic barriers are believed to be significant stressors on American eel (Machut et al., 2007) - lack of riverine habitat connectivity is believed to affect juvenile 'glass' and 'elver' stages of the American eel (Atlantic States Marine Fisheries Commission, 2000) and mortality is high for adult eels when passing downstream through hydropower turbines (COSEWIC, 2006a).

3.1.2 Barriers at Road Crossing in Nova Scotia

On a provincial scale, the extent of fragmentation of river systems due to barriers at road crossings is significant. The count of culverts extracted from NSTDB roads layer numbered over 39,000 (see Figure 1; GeoNova, 2012). A spatial analysis revealed an approximate 62,875 km of river and stream (excluding 'river-lake', 'lake', and 'coastal river'(GeoNova, 2012) in the province, and thus a culvert for every 1.6 km of stream/river. This is likely a conservative estimate - it is not clear from public records how many culverts, if any, are contributed to the provincial spatial dataset from the 'culvert notification process,' in which the NSE is notified that a culvert is being installed (Langill & Zamora, 2002). The metadata associated with the provincial dataset only cite photogrammetric and contracted surveyor sources for culvert locations (Access Nova Scotia, 2010).

Efforts have been made recently to locate and evaluate the impacts of culverts and prioritise their mitigation (Hynes et al., 2005; Hicks & Sullivan, 2008). Omitting barriers on non-fish-bearing streams, Hicks and Sullivan (2008) surveyed 60 barriers in Southwest Nova Scotia and found 33 (55%) of them to pose passage problems. In another study, a random sample of 50 culverts installed in 1999 and 2000 in Colchester, Cumberland, Halifax, and Hants Counties revealed that roughly 48% posed problems to fish passage (Langill & Zamora 2002).

Many river systems in the province developed for hydropower are presently home to or historically contained various species of diadromous and potamodromous migratory fish. To mitigate impact on fish passage, NSPI has installed fish ladders at a number of hydropower dams. There is continuing pressure by provincial regulators and community organizations to install additional mitigation structures at other dams around the province. To demonstrate the application of the models and methods presented in this research, three Nova Scotian river systems were selected: Mersey, Sheet Harbour (East River), and St. Margaret's Bay (see Figure 7). The three systems selected were chosen because they are actively managed for hydropower development, are home to current or historic populations of diadromous fish, and prioritising fish passage projects on these systems is a priority for NSPI (K. Meade, personal communication, March 18, 2010).

3.1.3 Mersey

The Mersey system is located approximately 120 km southwest of Halifax and is the largest system of the three chosen, with an approximate drainage area of 1963 km^2 (Figure 8; NSPI, 2010). The system was first developed for hydropower in 1903 by the town of Liverpool (NSPI, 2010). In 1928, the Nova Scotia Water Power Commission purchased the hydroelectric facilities (NSPI, 2010) and developed the Upper Lake Falls, Lower Lake Falls, Big Falls, and Jordan Lake dams (NSPI, 2009b). Currently, the system has a maximum generating capacity of 42 Megawatts (MW) and generates an average of 227.44 Gigawatt-hours (GWh) per year - approximately 23.6% of the total hydropower generation by NSPI (NSPI, 2010). Lake Rossignol (~129 km^2), located roughly in the centre of the system, acts as the main reservoir and is the biggest freshwater body in the province. The dams at Jordan Lake and Sixth Lake redirect drainage from the upper reaches of the adjacent Jordan River system into Lake Rossignol. Prior to flooding for hydroelectric development, eleven lakes existed on the current footprint of Lake

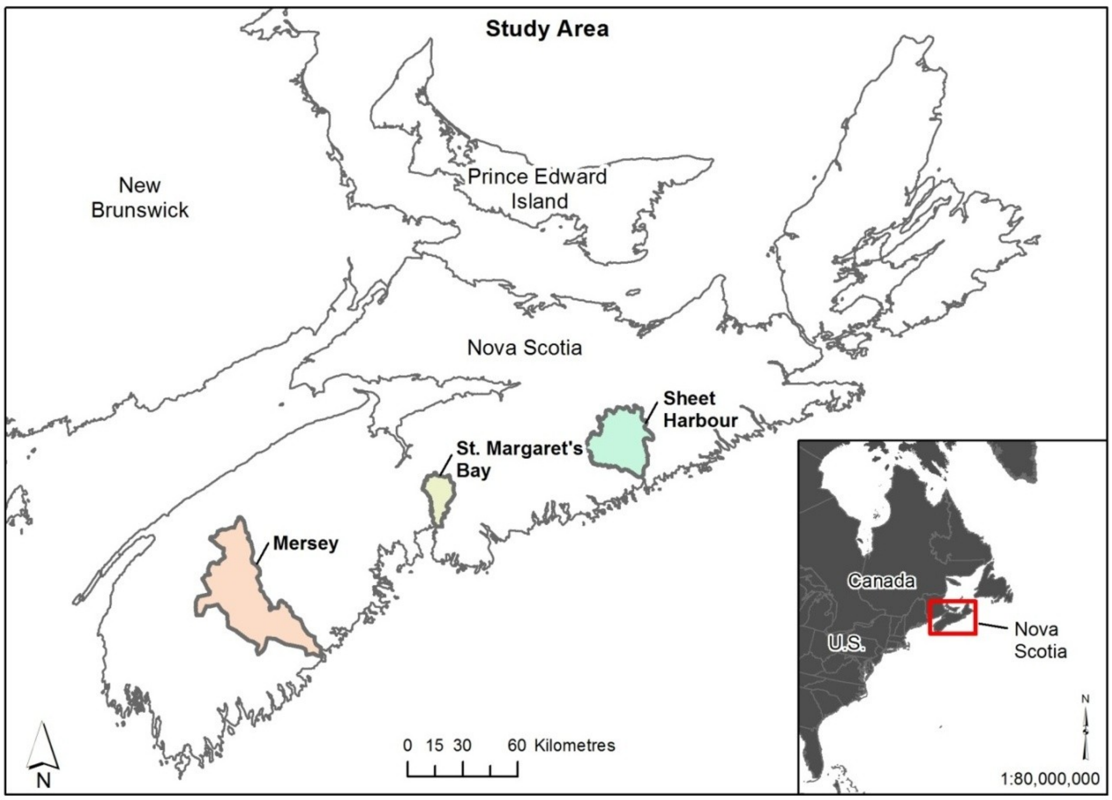

Figure 7: Study area showing three selected river systems.

Rossignol (Mersey Tobeatic Research Institute, 2009, in NSPI, 2010). Kejimkujik National Park (403 km^2; Geomatics Canada, 2012) is located upstream of Lake Rossignol.

Several fish species of special concern are currently present on the Mersey system. The Nova Scotia Salmon Association lists the Mersey as a location of a fall salmon run (Nova

Scotia Salmon Association, 2012), though NSPI reports that only one salmon was captured at the Cowie Falls fish ladderbetween 1997 and 1999 and none have been reported since (NSPI, 2010). In general, it is not clear how community assemblages of freshwater fish have been affected by hydropower development on the Mersey system.Prior to development, the Mersey may have been a seasonal home to a population of spawning Atlantic Salmon, though it is difficult to conclude how large the population was or how far upstream the fish would have been able to pass.

Two additional species exhibiting diadromous behaviour are known to be currently present: the alewife (*Alosapeusdoharengus*) and the American eel (*Anguilla rostrata*).A spawning population of alewife numbering approximately 10,000 fish is reported annually at the Cowie Falls fish ladder (NSPI, 2010). The American eel is the most ubiquitous and abundant species captured in monitoring surveys conducted by NSPI, with a wide variety of size classes present, suggesting successful recruitment (NSPI, 2010). It is unknown how juveniles ascend to upper reaches, past Lower Lake Falls and Big Falls; however, juvenile American eels are known to ascend barriers or venture out of the water to pass barriers (Legault, 1988; Haro & Krueger, 1991). It is also possible that they colonize Lake Rossignol and beyond via the Jordan River system where anecdotal accounts report large numbers of elver-stage (i.e. juvenile) eels ascending dams (NSPI, 2010). The brook trout is a potamodromous salmonid also confirmed to be present in the system (NSPI, 2010).

Fish ladders facilitating upstream fish passage have been installed at the first four dams of the system: Milton dam, Cowie Falls dam, Deep Brook dam, and Lower Great Brook dam (NSPI, 2009b). Three dams are found within the park boundary: Beaverskin Lake Dam, Little Peskowesk Lake Dam, and Hilchemakaar Lake Dam.These dams are estimated to be 50% passable by Kejimkujik biologists (D. Pouliot, personal communication, September 12, 2011). The Lower Lake Falls and Big Falls dams are the largest on the system, at 20.1 m and 15.9 m respectively (NSPI, 2009b). These dams do not allow fish passage (NSPI, 2009b).

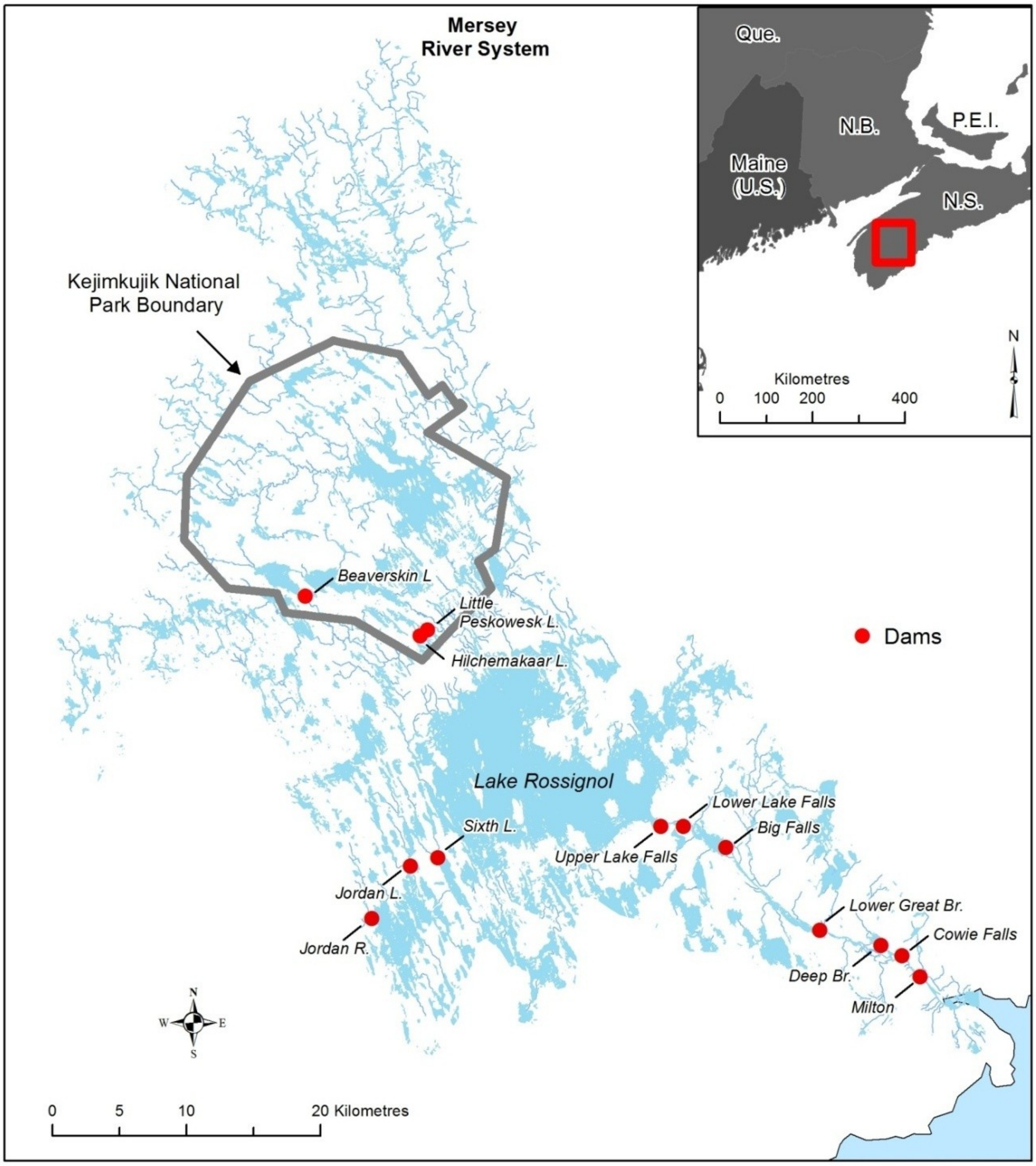

Figure 8: The Mersey river system. Rossignol, roughly at its centre, acts as a reservoir for Upper and Lower Lake Falls dams.There is potentially an inter-basin transfer to the Jordan River at the Jordan River Dam.

3.1.4 St. Margaret’s Bay

The St. Margaret's Bay system is located approximately 20 km northwest of Halifax and has a drainage area of approximately 271 km^2 (Figure 9). The system has been developed for hydropower for at least 150 years. The current combined capacity of the generating stations at Mill Lake, Sandy Lake, and Tidewater is 10 MW, averaging production of 26.71 GWh per year, representing approximately 2.8% of the average annual hydroelectric production of NSPI (NSPI, 2009c). The system combines drainage from two rivers that naturally drain to the ocean in close proximity to each another: the Northeast and Indian Rivers.Both systems were historically home to seasonal migratory populations of Atlantic salmon and gaspereau, though none exist today (J.M. Nicolas, personal communication, August 1, 2012).

According to The Nova Scotia Water Power Commission (1916, in NSPI, 2009c), the Northeast River once had a considerable reputation for salmon fishing. Speckled trout have been stocked in Mill Lake (Nova Scotia Department of Fisheries and Aquaculture, 2007, in NSPI, 2009c) and are present throughout the system (NSPI, 2009c). American eel is also reported throughout the system (Davis & Browne 1996; NSPI, 2009c).
The natural outflows of the Northeast and Indian Rivers are from Mill Lake in the east and Little Indian Lake in the west, respectively. At Little Indian Lake, there are diversion screens which redirect flow to Mill Lake through a diversion channel.These diversion screens pose a barrier to fish passage (J.M. Nicolas, personal communication, August 1, 2012). There is also a pipe diverting water from the upper end of Little Indian Lake to above the hydroelectric facility at Mill Lake.Currently, flow volume leaving the old channel of Little Indian Lake is low and would prevent fish movement (J.M. Nicolas, personal communication, August 1, 2012). Moving upstream in the eastern half of the system, there is Mill Lake Dam, Coon Pond Dam (generating), Wright's Lake Dam (storage), and Pockwock Lake Dam (Pockwock Lake is one of Halifax Regional Municipality's drinking water supply reservoir). Fish passage projects on the eastern side of the system are less likely to be undertaken by NSPI because Pockwock Dam is managed by the Halifax Regional Municipality and entrains a large portion of the watershed (J.M. Nicolas, personal communication, August 1, 2012). On the western side of the system, moving upstream, is the Little Indian Lake Dam (diversion), Sandy Lake Dam (generation), Big Indian Lake Dam (storage), and Five Mile Lake Dam (storage). In addition to the main dam at Five Mile Lake, the Mack dam and Beeswanger Dam are

present at the southwest and northeast of the lake, respectively.The Sandy Lake Dam poses the largest mitigation challenge, with a height of approximately 24.9 m (NSPI, 2009b). All other dams on the system are 10 m high or less (NSPI, 2009b). In summary, there is effectively no connectivity to the system from the ocean (NSPI, 2009c).

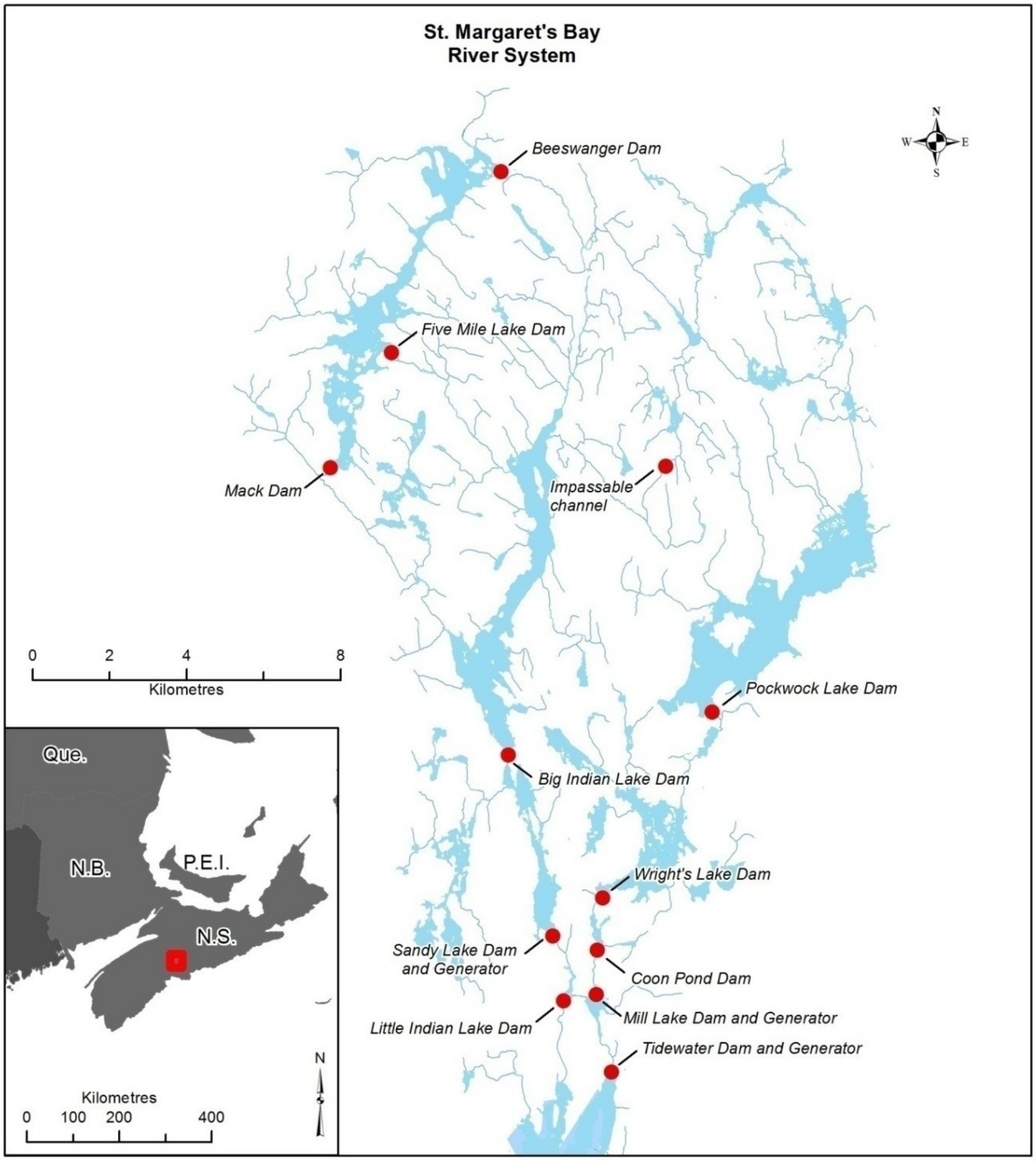

Figure 9: The St. Margaret's Bay river system, the combination of two watersheds, with a human-made diversion at Little Indian Lake to Mill Lake.Pockwock Lake is a drinking water reservoir of Halifax Regional Municipality.There is also a potential inter-watershed connection, labelled 'impassable channel.'

3.1.5 Sheet Harbour

The Sheet Harbour River system is located on the Eastern Shore of Nova Scotia, approximately 85 km northeast of Halifax (Figure 11). Two river systems drain to the ocean in close proximity: the East River and the West River.The East River system is used for hydroelectric generation and has a drainage area of approximately 570.6 km^2. This system currently generates 4.5% (43.31 GWh/yr) of the average annual production of hydropower by NSPI, with an installed capacity of 10.6 MW (NSPI, 2009a).

The East River has been developed for hydroelectric purposes for 89 years, with the Malay, Governor Lake, Anti, Sloan, and Ten Mile Lake dams completed between 1923 and 1924 (NSPI, 2009b). Prior to that, timber operators used the river for milling and transport since at least 1830 (Rutledge, 1954, in NSPI, 2009a). Reports of impacts of these operations to fish passage have been reported as far back as 1881 (NSPI, 2009a). Historically, the East River was particularly productive for Atlantic salmon.However, the development of the system for hydropower production is believed to have reduced the numbers of adult salmon able to access spawning grounds. A spawning population numbering approximately 200-300 remained until the 1960's when the stock entered a period of decline from which it has not recovered. Adult returns to the river between 2003 and 2007 have been fewer than five fish (NSPI, 2009a).

The American eel is found throughout the system. The exact mechanism of passage of the juvenile eel to the upper reaches is not known, though it is speculated that their ability to climb rough, wet surfaces or venture out of the stream aids in their passage. Downstream passage and consequent mortality through turbines is not fully understood (NSPI, 2009a).

The brook trout, a potamodromous fish, is present in the East River system.One of only two trout species native to Nova Scotia (along with the lake trout), it can tolerate salt water, an adaptation that allows it to use estuaries to move between river systems (Davis

& Browne, 1996). Brook trout were found throughout the East River system by NSPI surveys (NSPI, 2009a).

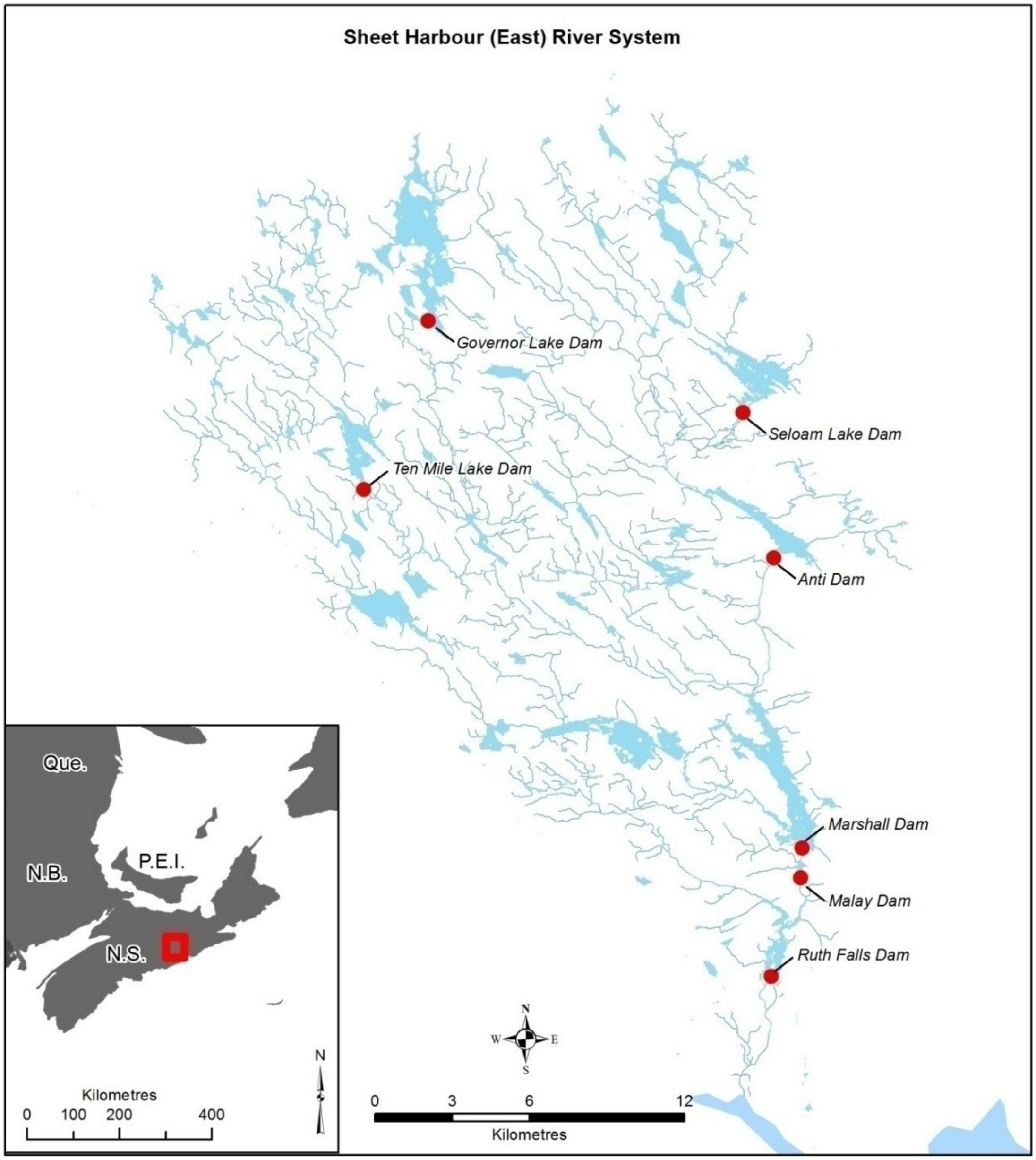

Figure 10: The Sheet Harbour system - limited to the East River for this study.

Current fish passage is limited mostly to downstream bypasses which are present on all dams except Governor and Sloan Lake dams. An upstream pool and weir fish ladder is installed at Ruth Falls.A trap–and-truck operation is used to transport fish from below the

Malay dam to above Marshall dam. The largest dam, at 10.4 m, is the Anti Dam which permits no upstream passage (NSPI, 2009a).

3.2 Network and Barrier Data

Geospatial data for river network features and barriers were acquired from various sources. Parameters such as barrier mitigation cost, the mitigation options available (e.g., downstream bypass, fish ladder, culvert replacement, etc.), whether or not a barrier actually posed a connectivity impediment, and barrier permeability were estimated using basic models and methods found in the literature, expert consultation, and existing data. The following section details these methods.

3.2.1 River Network Lines and Polygons

River network lines and polygons were downloaded for the three river systems from the NSTDB (1:10,000 accuracy; GeoNova 2012) as ESRI 'feature classes' (a file type; ESRI**,** 2012a). The lines and polygons both contained existing feature codes that conform to National Hydrographic Network and NSHN data standards and included attributes (Government of Nova Scotia, 2007). The network lines and polygons were assigned additional attributes and their positions were often moved or modified slightly. These edits were made during the process of geometric network creation, described in Section 3.3.

3.2.2 Dam Locations

Dam locations were acquired from a number of sources.First, the point features contained in the NSTDB hydrographic network point layer with a feature code beginning with 'WADM' were extracted (GeoNova, 2012). Second, a dam registry created by NSPI was cross-referenced with these points. If a dam described in the NSPI registry was absent from the NSTDB data, it was added to the list.Relevant dam attributes were also added (dam height, dam width, fish passage information, dam type) from the NSPI data. Third, dam locations within Kejimkujik National Park, Mersey system, were acquired from Parks Canada.Consultation with staff of Parks Canada (D. Pouliot, personal communication, August 20, 2011) and NSPI (D. Thompson, personal communication, May 22, 2012) was done to discuss three barriers listed in the NSTDB but not associated with NSPI operations or located inside Kejimkujik National Park boundaries. These barriers were determined completely passable and removed from consideration.A total of 36 dams were left in the dataset for the three systems. Further discussion with NSPI staff (D. Thompson, personal communication, May 22, 2012) was done to determine that 13 of the 36 dams were structures adjacent to or associated with another dam or were not on a waterway (e.g., a 'wing' dam) leaving 23 dams for use in the analysis (Table 3).

Table 3: Dams with Barrier Permeability Values and Estimated Costs and Benefits of Repair Projects.

#	Dam name	River System	Fishway Type	Perm-eability	Option 1: Project Type	Option 1: Pass. After	Option 1: Cost ($000s)	Option 2: Project Type	Option 2: Pass. After	Option 2: Cost ($000s)	Option 3: Project Type	Option 3: Pass. After	Option 3: Cost ($000s)
1	Jordan Lake	Mersey		0.0	US	0.5	59	DS	0.5	59	US & DS	1.0	118
2	Milton Roll	Mersey	Variable upstream passage	0.3	US	0.5	196						
3	Cowie Falls	Mersey	Pool & weir concrete upstream	0.5	DS	1.0	662						
4	Deep Brook	Mersey	Pool & weir concrete upstream	0.5	DS	1.0	957						
5	Lower Great Brook	Mersey	Pool & weir concrete upstream	0.5	DS	1.0	498						
6	Big Falls	Mersey	No passage	0.0	US	0.5	898	DS	0.5	898	US & DS	1.0	1797
7	Upper Lake Falls	Mersey	No passage	0.0	US	0.5	1043	DS	0.5	1043	US & DS	1.0	2086
8	Lower Lake Falls	Mersey	No passage	0.0	US	0.5	1318	DS	0.5	1318	US & DS	1.0	2637
9	Jordan Lake	Mersey	No passage	0.0	US	0.5	65	DS	0.5	65	US & DS	1.0	131
10	Beaverskin Lake	Mersey	Downstream passage present	0.5	US	1.0	50						
11	Hilchemakaar Lake	Mersey	Downstream passage present	0.5	US	1.0	100						
12	Little Peskowesk Lake	Mersey	Downstream passage present	0.5	US	1.0	100						
13	Marshall	Sheet Harb.	Downstream bypass	0.5	US	1.0	400						
14	Ruth Falls	Sheet Harb.	Pool & weirupstream; louver & downstream bypass	1.0		0.0	642						
15	Malay	Sheet Harb.	Two downstream bypasses	0.5	US	1.0	498						
16	Governor Lake	Sheet Harb.	No passage	0.0	US	0.5	203	DS	0.5	203	US & DS	1.0	406
17	Seloam Lake	Sheet Harb.	No passage	0.0	US	0.5	203	DS	0.5	203	US & DS	1.0	406
18	Anti	Sheet Harb.	Downstream bypass	0.5	US	1.0	682						
19	Ten Mile Lake	Sheet Harb.	Downstream bypass	0.5	US	1.0	380						
20	Little Indian	St. Marg. Bay	No passage	0.0	US	0.5	30	DS	0.5	30	US & DS	1.0	60
20	Sandy Lake	St. Marg. Bay	No passage	0.0	US	0.5	1633	DS	0.5	1633	US & DS	1.0	3266
21	Big Indian Lake	St. Marg. Bay	No passage	0.0	US	0.5	662	DS	0.5	662	US & DS	1.0	1325
22	Five Mile Lake	St. Marg. Bay	No passage	0.0	US	0.5	400	DS	0.5	400	US & DS	1.0	800
23	Impass. Channel	St. Marg. Bay	No passage (Note: interbasin trans.)	0.0	US & DS	1.0	300						

3.2.3 Dam Permeability Estimation

Estimates of dam permeabilities of were made using consultation with NSPI staff, Parks Canada staff, and past studies.The permeabilities of dams owned and operated by NSPI were estimated based on judgment of wildlife biologists and environmental specialists working for the company (D. Thompson, personal communication, May 22, 2012) combined with the presence/absence of existing mitigation measures (e.g., fish ladders, diversion screens). Of the 36 identified structures associated with NSPI operations, 14 were omitted from analysis because they did not obstruct longitudinal connectivity (e.g., a 'wing' dam) or they existed on a braided section in parallel with another structure.In the latter case, only one barrier in a braided section was considered (see Figure 11) with the main structure taking priority. Of the remaining 22 structures, nine had fish passage measures installed. Of these nine, four had downstream bypasses only, four had upstream measures only, and one had both upstream and downstream measures present (Table 3). A single permeability estimate for each barrier was made by weighting the upstream and downstream permeabilitiesequally with each comprising 50% of the permeability index, thus assigning a simple average of the upstream and downstream permeabilities to each barrier.Permeabilityestimates of the three dams present inside Kejimkujik National Park were made by Parks Canada staff using the Fish Xing software and methods (Washington Department of Fish and Wildlife, 2006).

3.2.4 Dam Mitigation Options and Cost Estimation

Cost estimates for each barrier were made in consultation with managers of NSPI and an industry formula based on the assumption that costs of repair are a function of the height of the dam (Connecticut River Watershed Council Inc, 2000; Rhode Island Habitat Restoration Portal, 2003; K. Meade, personal communication, March 18, 2010; see Section 2.11):

$$y = 3.28x \times 20{,}000 \tag{3.1}$$

where y is the cost of the project and x is dam height.

There was a maximum of three mitigation options considered at each barrier: upstream fishway, downstream fishway/bypass, and both. Both downstream and upstream costs were estimated using the same cost formula. An upstream or downstream project was expected to improve permeability up to a maximum of 0.5. That is, a barrier with a functional upstream fishway and no downstream passage would have a maximum permeability of 0.5. If a downstream repair option was considered at that barrier, it was assumed to improve permeability from 0.5 to 1.0. The estimates of costs of full repair of dams ranged from 50,000 CAD at Beaverskin Lake dam in Kejimkujik National Park to 2.6 million CAD at Lower Lake Falls dam, both on the Mersey system.

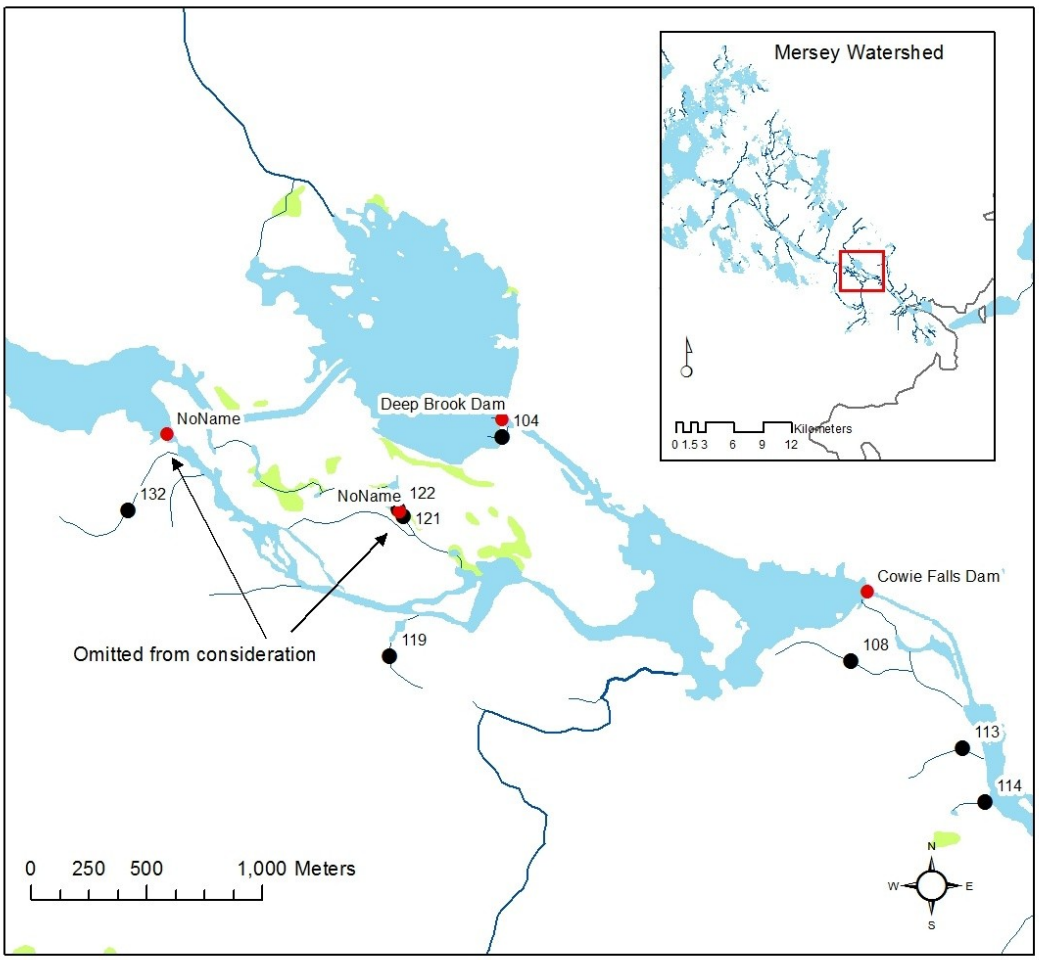

Figure 11: A braided section on the Mersey system.Network flow is disabled at the two 'NoName' structures.This allows the network to retain its tree-like structure and forces network flow through Deep Brook dam.

3.2.5 Culvert Locations

Culvert locations were extracted from the NSTDB road lines layer (Layer: 'RRLine' Code: 'RRCL50'). The midpoints of these lines were found using the ArcGIS 'Feature to Points' tool. These points were snapped to the river network lines after geometric network creation (see Section
3.3.2 Geometric Network Creation). Any duplicate points were removed in cases where multiple culverts were found at the same road crossing. A total of 181 culverts in the Mersey system, 250 in the Sheet Harbour system, and 125 in St. Margaret's Bay system were located and used.

3.2.6 Culvert Permeability Estimation

A permeability of 50% was assigned to all culverts. Existing models for estimating culvert permeability from available geospatial data are lacking, and the time commitment required and complexity involved in existing models which use data collected from site surveys (e.g.,Washington Department of Fish and Wildlife**,** 2006) was not within the scope of this research. One local study surveyed 60 culverts and found 33 (55%) impaired fish passage (Hicks & Sullivan, 2005). Although 50% of culverts in Nova Scotia may impair connectivity, it is difficult to ascertain which ones without site visits (see Section 2.10). Due to the lack of information, an educated guess was ultimately made that estimated all culverts to be 50% passable bidirectionally.

3.2.7 Culvert Mitigation Option and Cost Estimation

All culverts costs were estimated at 15,000 CAD. The costs associated with culvert remediation are highly dependent on information only available through *in situ* site assessments; no models could be found that link available watershed-level geospatial data to culvert mitigation / repair costs (see Section 2.11). A range of average culvert costs between 10,000 CAD (Parker, 1999) and 100,000 CAD (Fish Passage Technical Working Group, 2012) was found in the literature. A manager at NSPI estimated local culvert project costs at the low end to be 10,000-15,000 CAD (K.Meade, personal communication, January 17, 2011). A survey of 37,000 fish passage projects in the U. S. found the median project cost to be 30,000 USD (Bernhardt et al., 2005). An estimate closer to the lower end of the project cost spectrum was made because the NSPI estimate was made with local knowledge. Culvert mitigation options were assumed identical at all culverts and to repair connectivity at each culvert to 100%.

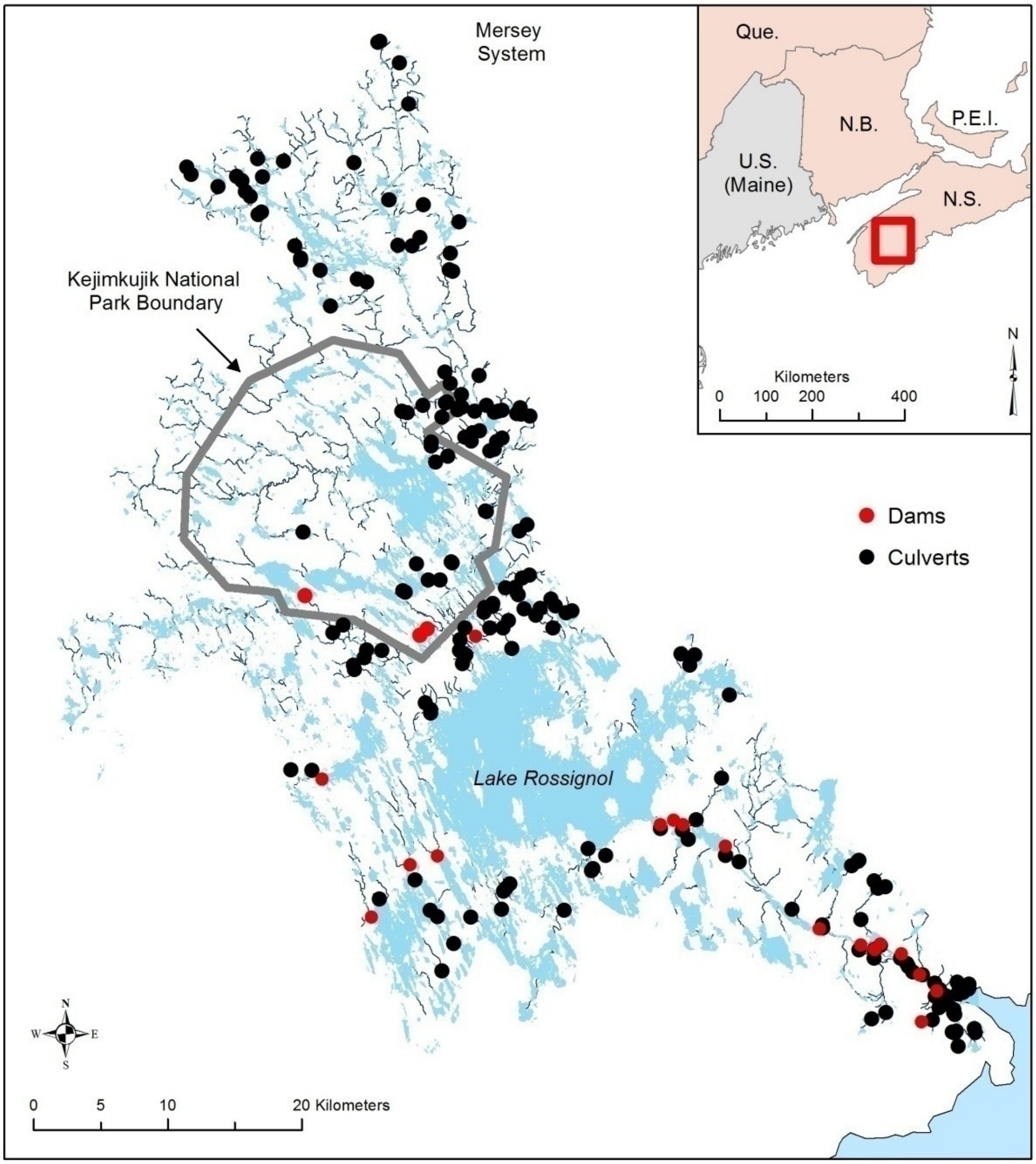

Figure 12: The Mersey system, with 181 culverts and eight dams included in analyses.

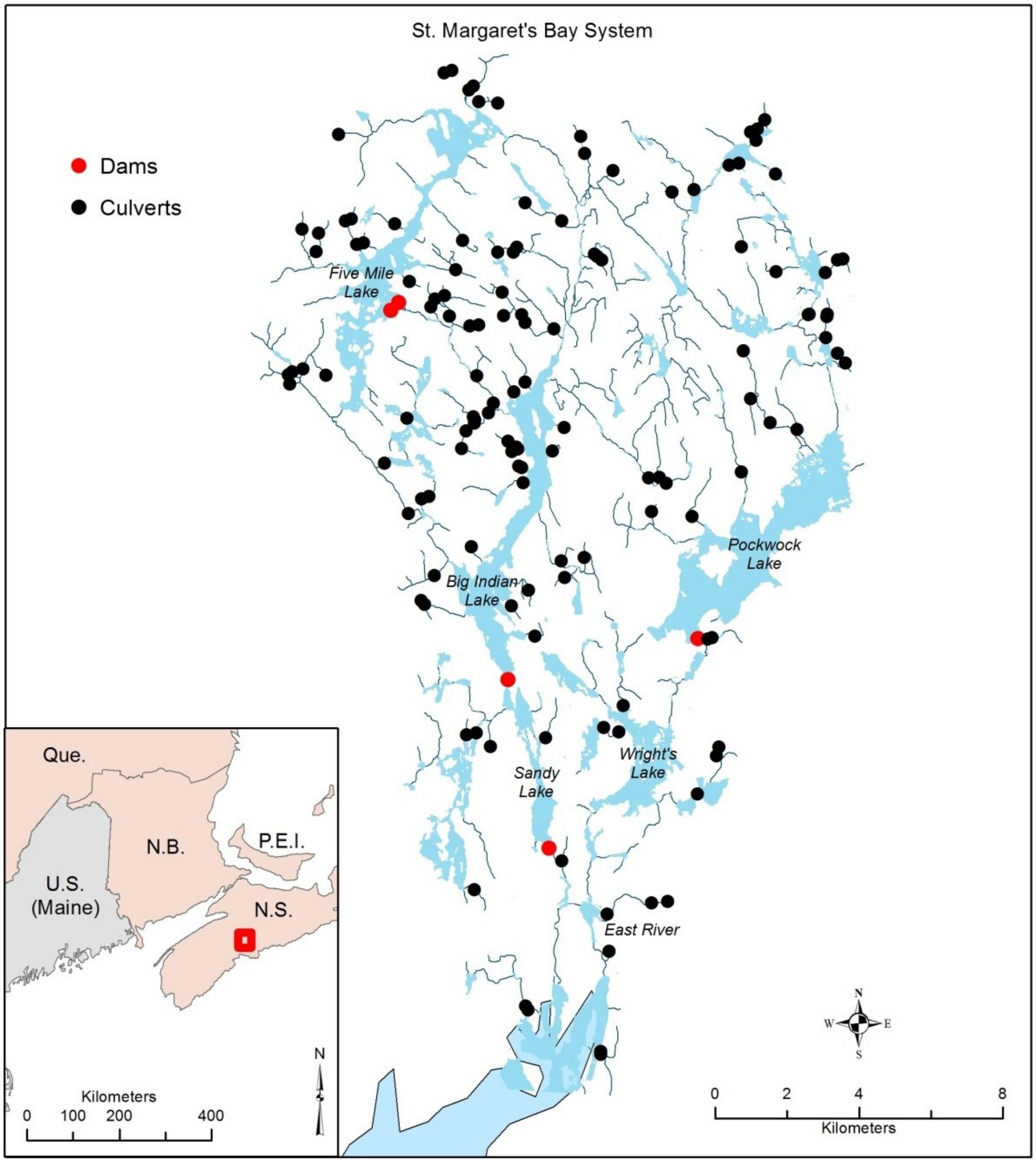

Figure 13: The St. Margaret's Bay system, with 125 culverts and nine dams included in analyses.

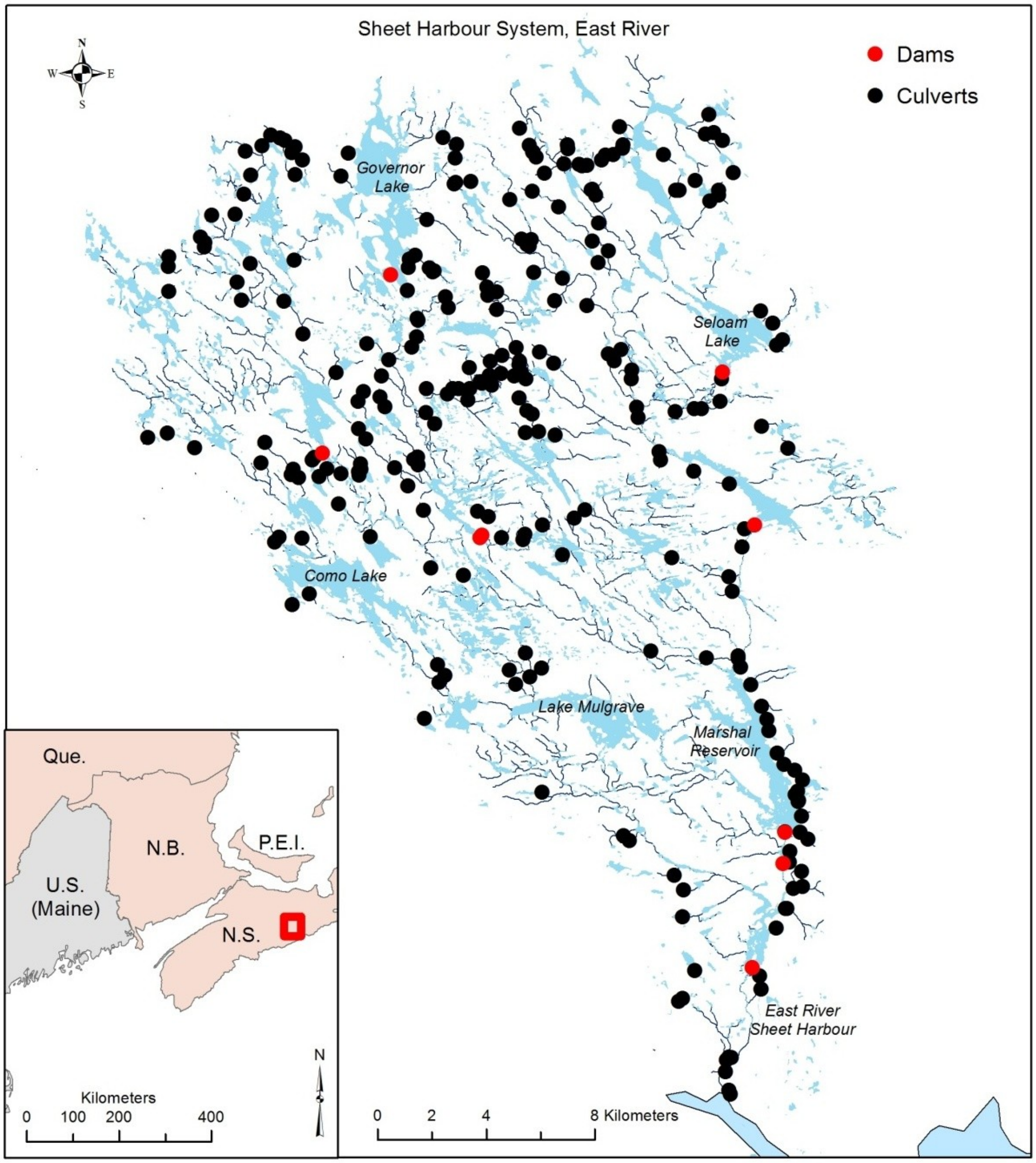

Figure 14: The Sheet Harbour (East River) system, with 250 culverts and six dams included in analyses.

3.3 Data Preparation

3.3.1 Watershed delineation

Geopatial watershed layers from NSPI and Fisheries and Oceans Canada were used as the basis for watershed delineation. The source of watershed delineations was a 20m digital elevation model of the province. The layers refined by NSPI to include only water-bodies connected to systems managed for hydroelectric operations. Watersheds were refined again by excluding the West river of Sheet Harbour and the Jordan River system (connecting to Jordan Lake and subsequently to the Mersey system).

In the case of the Mersey system, geospatial layers indicated the Jordan River watershed to be connected to the Mersey system near the west end of Lake Rossignol.Consultation with NSPI revealed that Jordan Lake and Sixth Lake at the headwaters of the Jordan system have been converted to storage reservoirs, with flow diverted into Lake Rossignol (D. Thompson, personal communication, May 22, 2012). Historically, these lakes would have been fed from Lake Rossignol but flow direction in the connecting tributary was reversed in 1929 (or earlier) via the installation of Jordan Lake dams, forcing flow into Sixth Lake and then into Lake Rossignol (NSPI, 2010).Neither Sixth Lake nor Jordan Lake dams have fish passage (D. Thompson, personal communication, May 22, 2012). The connectivity between the two systems at this location means that there are two potential routes between the ocean and Lake Rossignol, creating a loop in the network. The directed model requirement of a single route to and from the oceanled to the network beingdisconnected at the Jordan Lake dam (**Figure 15**), restricting access to the ocean to via the Mersey system. This disconnection is believed to represent the reality of the situation – no fish passage exists connecting the Mersey and Jordan River systems – but future studies may consider looking at fish passage projects at Jordan lake as a means of providing connectivity from the oceanthrough to Lake Rossignol and beyond.

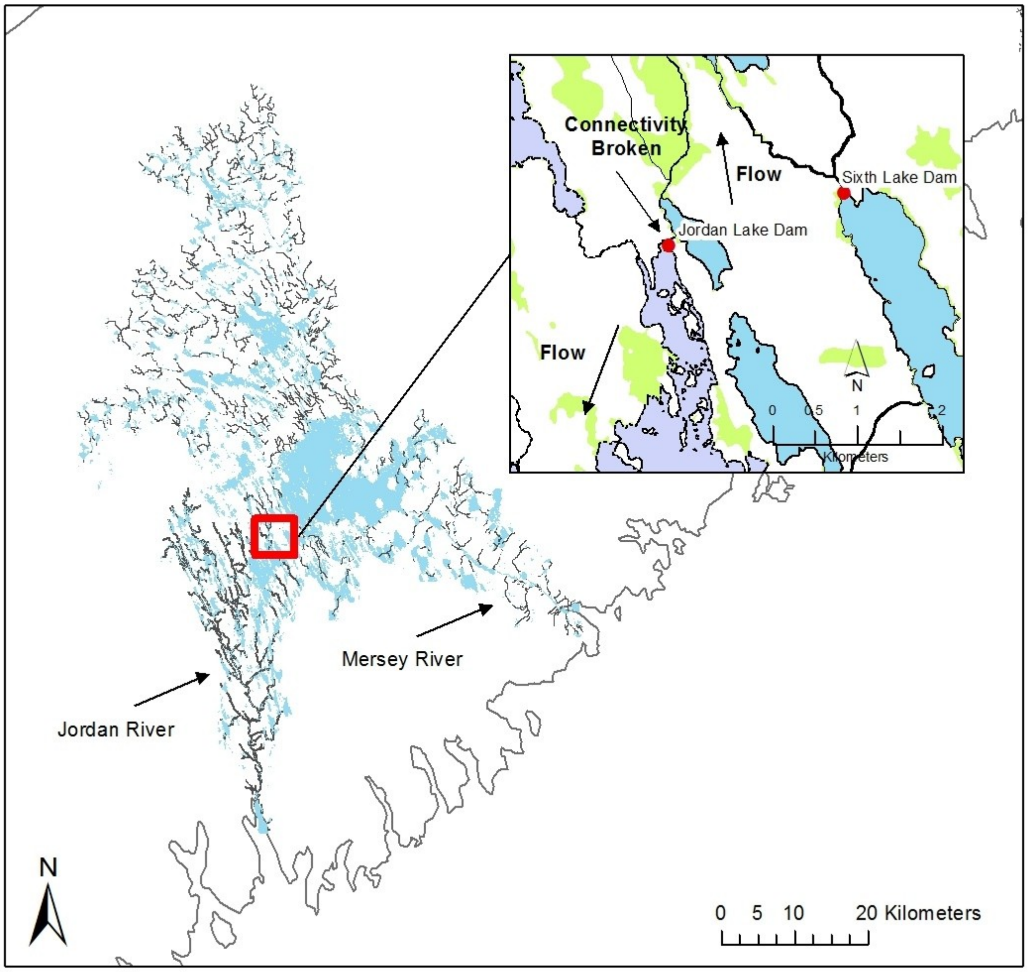

Figure 15: The inter-basin transfer between the Jordan and Mersey system at Jordan Lake. This connectivity is impaired by the Jordan Lake Dam and was considered 100% impassable (with no repair options) during analyses.

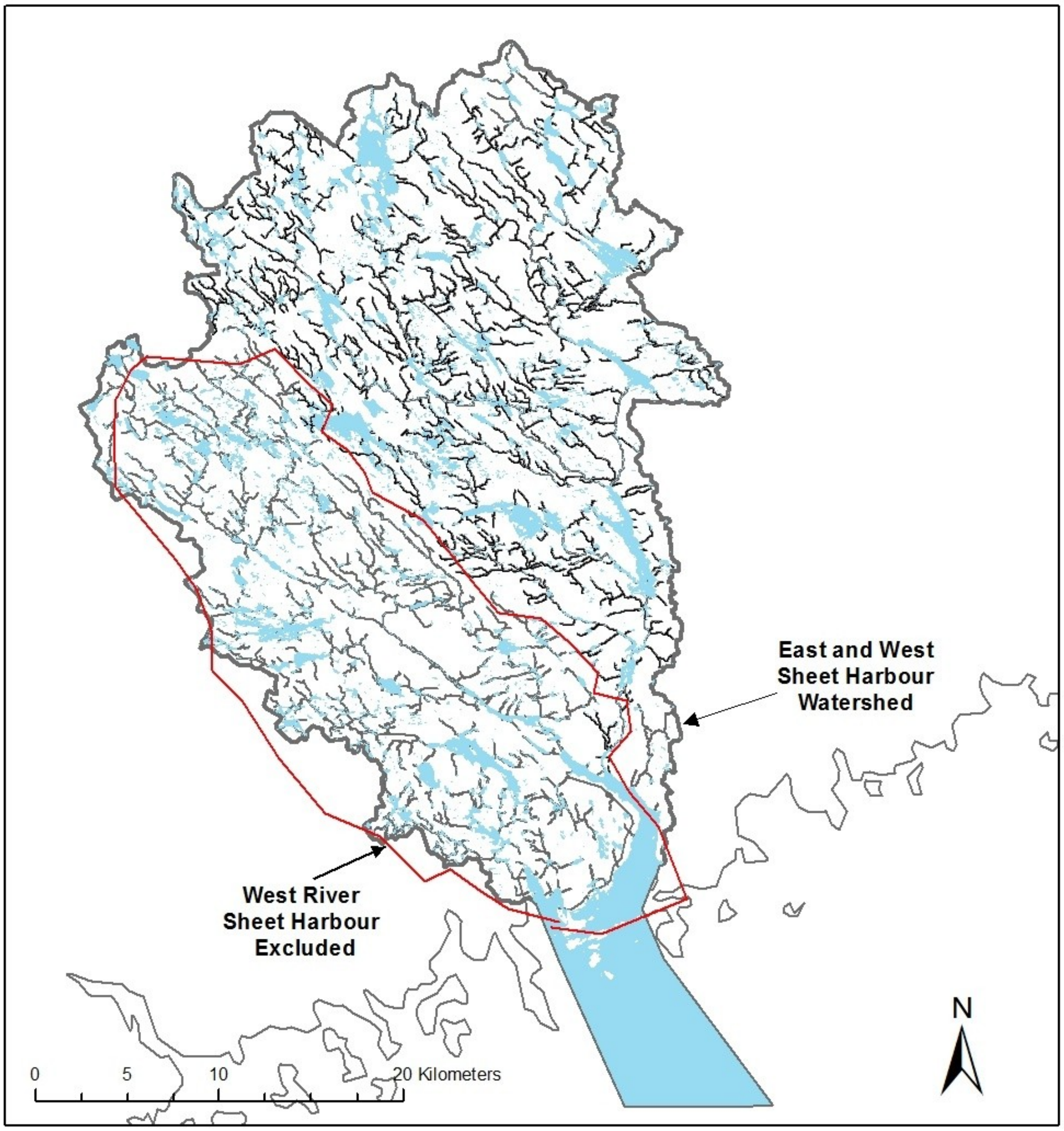

Figure 16: The East and West River of the Sheet Harbour system.

For the remaining two systems, SheetHarbour and St. Margaret's Bay, minor network edits were made. For the Sheet Harbour system, the West River was excluded from analysis because only the East River of Sheet Harbour is actively managed for hydroelectric generation (Figure 16). For the St. Margaret's Bay System, the eastern outflow through the Tidewater generator was eliminated from analysis after consultation revealed that an inter-watershed connection happens because of a built channel between Mill Lake and Little Indian Lake and that no fish passage projects would be considered between Tidewater dam and Mill Lake.Projects at the Pockwock Lake Dam, which currently does not have fish passage and is considered 100% impassable, were also excluded from analysis because Pockwock Lake is used as a drinking water reservoir for the city of Halifax and passage-relatedprojects would be unlikely. Examination of the GIS layers revealed that an inter-watershed cross-over could occur at a channel south of Clement's Lake, connecting the east and west sides of the systems. Consultation with NSPI managers revealed that this channel is likely impassable due to low flow volume (J.M. Nicolas,personal communication, August 1, 2012). A project at this point, which would connect the East River watershed above Pockwock Lake to the Indian River watershed, *was* considered in the analyses of this research.

3.3.2 Geometric Network Creation

Geometric networks of all three systems were created using ArcGIS Desktop (ESRI, 2012a). The geometric network model is bundled with the ArcGIS Desktop suite of software along with a toolset for analyzing electrical and water distribution networks, called Utility Network Analyst (ESRI, 2012b). First, river line layers and barrier point layers were imported as 'Feature Classes' (a file type) within a 'Feature Dataset' (similar to a windows folder) of a 'File Geodatabase' (a compressed database format provided with ArcGIS). A layer was created to hold points designating sinks of the network. A geometric network was then built using 'simple edges', no 'weights', and no 'm-values' (i.e., routing values). During network creation, barriers were snapped (i.e., moved to lines) up to 50 m if needed. After network creation, outflow points were created manually in the 'sinks layer' for all networks. Flow direction was then set using the 'set flow direction' button, which calculates flow direction based on network topology (as opposed to elevation or digitized direction).

A significant amount of time was taken to inspect river lines and barriers for each network. A number of common errors were encountered:

- despite a 'snapping' feature in the geometric network build process, barriers were not snapped or connected to lines after network creation
- duplicate points existed at culvert locations

- line segments were disconnected from the network
- inter-basin transfers occurred (multiple sinks or outflow points)
- large braided sections or looped sections caused indeterminate stream-flow direction

The steps taken to deal with these issues are outlined in (see Appendix B.1).

3.4 Optimisation Models

3.4.1 Overview

Two mixed integer linear programs were developed for optimising barrier removal. The first was created to solve the problem diversely described in the literature as *maximising upstream-downstream connectivity* (sensu Kuby et al., 2005), *upstream-downstream accessibility relationships* (sensu Zheng et al., 2009), the *fish passage barrier removal problem* (O'Hanley & Tomberlin 2005), or diadromous connectivity (Cote et al., 2009). This can be summarized as a problem of maximising habitat accessibility or connectivity for diadromous fish to and from the ocean. However, I will refer to this problem as one of *maximising directed longitudinal connectivity* to keep it consistent with network theory terminology (see Proulx et al., 2005) and more broadly applicable to systemic connectivity. The second model described here is created to solve the problem of *maximising undirected longitudinal connectivity*, that is, the connectivity of the system regardless of flow direction. This type of connectivity can be quantified with the potamodromous DCI (DCI_p; Cote et al., 2009). To date, only O'Hanley (2011) has presented an optimisation model to maximise this type of connectivity, which they refer to as a *maximum edge-weighted connected subgraph problem*.

The model formulations presented here are 'barrier centric' in the sense that habitat is always associated with a barrier. This is in contrast to O'Hanley (2011) in which a 'segment centric' approach is used, though this distinction is only relevant for scripting and mathematical formulation and does not affect outcomes. The approach taken here considers network quantity and permeability as attributes of *barriers* with network quantity aggregated from network immediately upstream until the next barrier(s) or the headwaters. The choice is somewhat arbitrary; a network segment could have its associated network quantity along with its most downstream barrier associated with it. The use of the barrier-centric approach is made for ease of integration with the GIS toolset used to generate network data.

The assumption of a dendritic network structure is also important. One consequence of this assumption is that there is only one path between two points in the network (see Section
2.8 GIS and Dendritic Ecological Networks). For the model presented here, the main consequence is if there are braided sections in the network that create 'cross-overs' between barriers, then this will affect accuracy of network quantity calculations and subsequent prioritisation results. Specifically, the common network above two sibling barriers will be double-counted as network area (Figure 17). To prevent this, the river networks were edited to ensure that no significant braiding occurred (see Section 3.3.2).

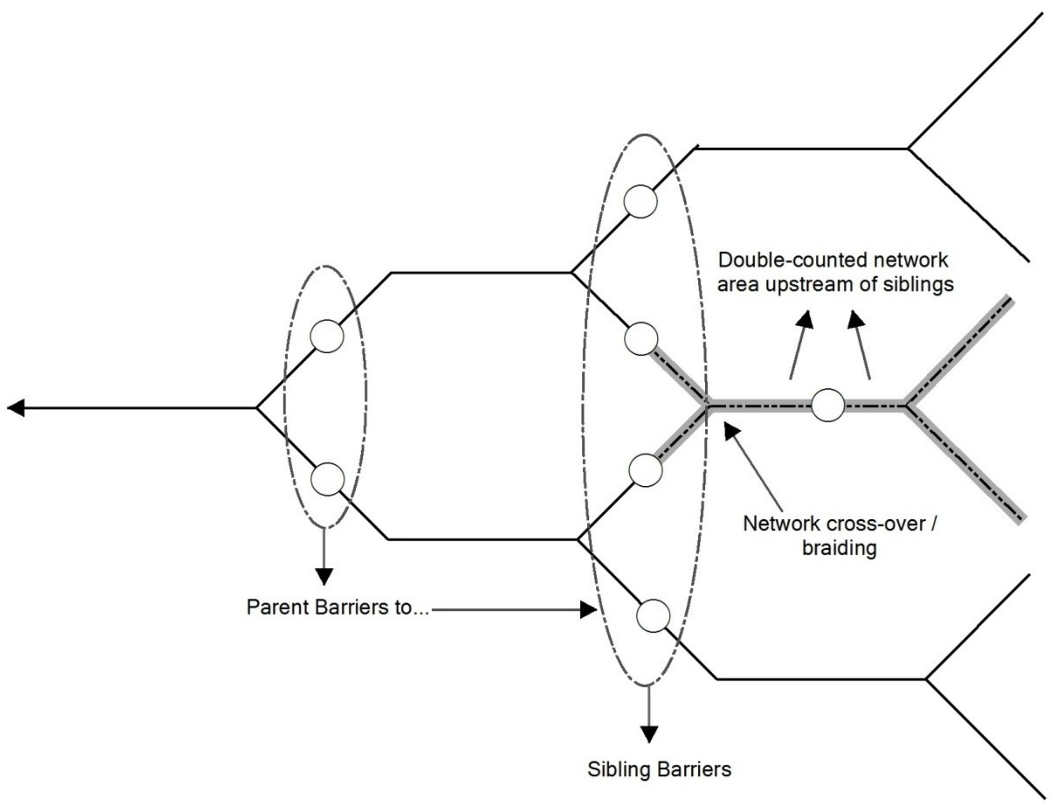

Figure 17: Illustration of braiding breaking dendritic network structure. This can lead to double-counting by network analysis algorithm of network quantity above sibling barriers.

Both models presented here are linear programs that incorporate continous permeabilities. Linear formulations are advantageous over non-linear ones for a number of reasons (see Section 2.9). Non-binary permeabilities are desirable because they do not impose the limitation of considering barriers as either ‘passable’ or ‘non-passable’ (see Section 2.10). For example, in situations where weighting relative benefits to different species are desired, non-binary permeabilities allow for simple weightings to be applied to permeability improvements between species.

3.4.2 ‘Directed’ Model

Consider the following notation for maximising the permeability-weighted river network accessible to and from the ocean or network sink. It can be assumed that the set of barriers I are indexed by i and are all impairing longitudinal connectivity to some degree. The network upstream of any barrier y_i is denoted H_i. At each barrier there is a set of options O, indexed by k, each of which has a cost c_{ik}. The options at each barrier are assumed to include a 'do nothing' option which costs nothing and leaves the permeability of that barrier, denoted by p_i, unchanged. The permeability of each barrier is assumed to be the product of the upstream and downstream permeabilities. Assuming each barrier has potentially many upstream barriers and exactly one downstream barrier, the set of upstream barriers from a barrier i is denoted $U(i)$, indexed by j. The total budget is denoted by β. The following decision variable is used:

$$x_{ik} = \begin{cases} 1 \; if \;\; option \; k \; at \; barrier \; i \; is \; chosen \\ 0 \; otherwise \end{cases}$$

The first model for maximising directed longitudinal connectivity is described as follows:

Objective:

$$\text{Maximize } y_0 \tag{3.2}$$

Subject to the following constraints:

$$y_i = \sum_{k \in O(i)} z_{ik} \qquad \forall\, i \in I \tag{3.3}$$

$$z_{ik} \le \sum_{j \in U(i)} p_{ik} y_j + p_{ik} H_i \quad \forall\, i \in I, k \in O(i) \tag{3.4}$$

$$z_{ik} \le z_{ik}^{max}\, x_{ik} \qquad \forall\, i \in I, k \in O(i) \tag{3.5}$$

$$\sum_{k \in O(i)} x_{ik} = 1 \qquad \forall\, i \in I \tag{3.6}$$

$$\sum_{i \in I} \sum_{k \in O(i)} c_{ik} x_{ik} \le \beta \tag{3.7}$$

where:

y_0 = accessible network upstream of the system sink
I = the set of all barriers
i = a single barrier in the set of all barriers
O_i = the set of optionsat barrier i
k = a single option in the set of options
U(i) = the barrier(s) immediately upstream of i
H = the network immediately upstream of a barrier
j = a single barrier in the set of upstream barriers
y = optimised network upstream
z = accessible network upstream if an option is chosen
x = a binary decision variable
c = the cost of a repair option
β = the total budget

The directed model is a linear program with the objective of maximising the largest directed permeability-weighted subnetwork upstream of the network sink (i.e., outflow)

y_0 (3.2). Constraint (3.3) defines the accessible amount of network upstream of any given barrier i, if option k is chosen, or z_{ik}. Inequality (3.4) both constrains and defines the accessible network amount above i if option k is chosen as equal to or less than the sum of the permeability-weighted habitat for all barriers upstream ($\sum_{\boldsymbol{j}\in\boldsymbol{U}(\boldsymbol{i})} p_{ik}\, y_j$) plus the accessible network immediately above barrier i, $p_{ik}\, H_i$. Combined, (3.3) and (3.4) yield all permeability-weighted network available upstream from barrier i. Inequality (3.5) is the basic connection between the choice of option k and the habitat z_{ik} due to choosing that option; if x_{ik} is 0 then so is z_{ik}. The maximum possible network upstream is constrained in eqn. (3.5) to z^{max}. Contraint (3.6) limits the number of decisions at each barrier to exactly one and prevents 'partial' projects. The selection of options are constrained by the total budget in (3.7). This model was created for the GLPK as a .mod file (Makhorin, 2012; Appendix A.1). As noted in Section 2.1, scaling y_0 to the total network available upstream would yield the DCI_d metric (i.e., $\frac{y_0}{y_{total}} * 100$), depending on the definition of permeability adopted and method used to calculate it.

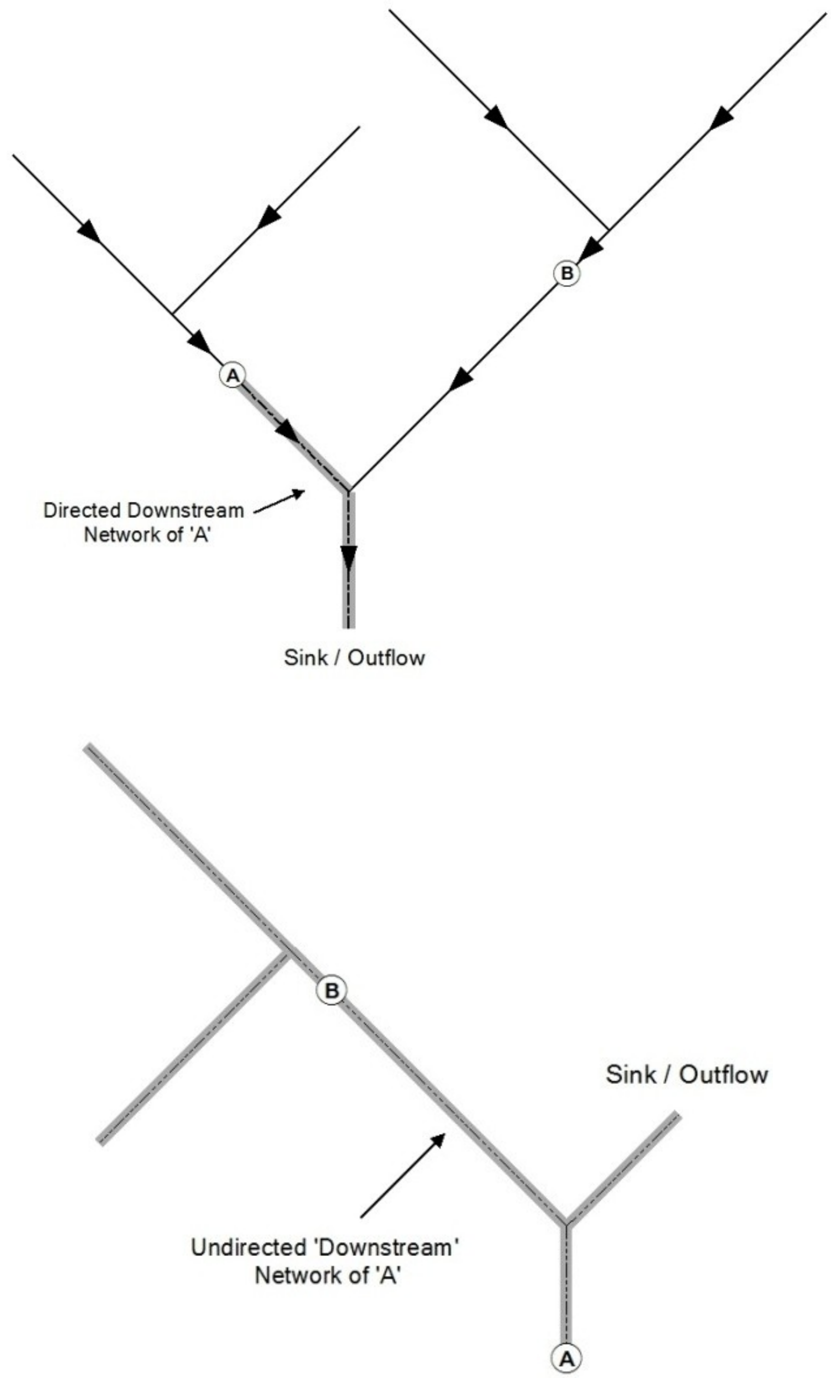

Figure 18: Directed vs. undirected connectivity differs in that the latter ignores the directional component of water flow in the system. The undirected downstream network of barrier 'A' can therefore also be viewed as a 'tree' which branches beyond barrier 'B'.

3.4.3 'Undirected' Model

To maximise undirected longitudinal connectivity, a similar approach can be taken. The undirected model has a similar objected to the program presented in O'Hanley (2011) and O'Hanley et al. (2013) and aims to maximise the *single largest undirected sub-network*. The problem of optimising for directed connectivity is a sub-problem of solving for undirected connectivity. Each barrier in the network is conceptualized as an outflow of both its connected upstream and downstream sub-networks; the barrier is the centre of two tree-like (i.e., dendritic) networks upstream and downstream (Figure 18: Directed vs. undirected connectivity differs in that the latter ignores the directional component of water flow in the system. The undirected downstream network of barrier 'A' can therefore also be viewed as a 'tree' which branches beyond barrier 'B'.).

To formulate the undirected model, consider the following notation in addition to what was defined for the directed model. Let the central barrier i to a given undirected subnetwork be defined as the single barrier downstream of the corresponding central river segment H_i. Let us assume for the moment that there are *many* barriers encountered 'downstream' from barrier i, denoted as a set by $D(i)$ and indexed by m. The term 'downstream' is thus applied loosely; all barriers in the downstream set are not necessarily downstream as defined by the flow of the river. Rather, they are the first barriers encountered in the subnetwork found in the downstream direction from barrier i (Figure 20). Note dendricity is still assumed. The network segment immediately downstream from a central barrier i can be given by H_m (i.e., the network upstream of the single downstream barrier m, following the flow of the river). Let the permeability-weighted accessible network found in the downstream direction from the central subnetwork H_i be denoted by q_i. Finally, the following additional decision variable is included:

$$\alpha_i = \begin{cases} 1 \; if \; barrier \; i \; is \; the \; central \; barrier \; to \; the \; maximal \; subnetwork \\ 0 \; otherwise \end{cases}$$

The upstream network accessible thru a given barrier z_i is calculated as it was in (3.4) of the directed model, but here the downstream accessible habitat is also required. The permeability-weighted accessible network downstream of a given barrier i is thus:

$$q_i = p_i H_m + \sum_{m \in D(i)} p_i w_m \quad i \in I \tag{3.8}$$

As in the directed model, the total network quantity downstream of i is defined by one contraint and one inequality. The partner constraint to (3.8) is thus the equivalent to constraint (3.3) which defines the optimal habitat downstream w_i:

$$w_i = \sum_{k \in O} q_{ik} \quad i \in I \tag{3.9}$$

The total accessible network through barrier i in both directions is thus:

$$y_i + w_i \tag{3.10}$$

In the undirected model, however, the 'centre' of the maximal sub-network must be a river segment, denoted H_i, and should not be weighted by permeability. The permeability-weighted H_i is calculated in (3.8) but must then be adjusted later to 'un-weight' it. 'Un-weighting'the network immediately upstream of the barrier, given by H_i, from the permeability, the sub-network quantity connected to the segment above barrier i becomes:

$$y_i + w_i - p_{ik} H_i + H_i \tag{3.11}$$

To determine the set of barriers immediately downstream of a barrier *D(i)*, more than one method could be employed. The connectivity matrix generated by the GIS toolset (discussed in Section

3.7 Integration of Optimisation Models with an SDSS) that defines network connectivity upstream from the network sink could be transformed; a sub-network downstream of barrier i could be extracted using a matrix transformation or with a simple algorithm. This could be performed 'on-the-fly' in the optimisation model or pre-calculated by the GIS toolset for all barriers (all i in I). Another approach was used here to avoid matrix transformations: to calculate permeability-weighted network downstream, the total network upstream of the single immediately downstream barrier *m* following network flow is found.

Let us now assume that the number of barriers in the set downstream *D(i)* from the central barrier i is restricted to one, and thus follows the flow of water in the network. Again, let us assume the network is dendritic. Let us also denote the set of barriers upstream from *m* as *U(m)* and be indexed by *j*. The calculation of downstream accessible habitat is thus reformulated as:

$$q_{ik} = p_{ik} H_m + \sum_{j \in U(m)} p_{ik} y_j - z_{ik} + p_{ik} w_m \quad i \in I, k \in O(i), m \in D(i) \tag{3.12}$$

In (3.12), the habitat downstream of the central barrier in a subnetwork *i* is weighted by the permeability at *i* and is thus $p_{ik}H_m$. The sum of all permeability-weighted habitat upstream of the immediate downstream barrier *m* is then found ($\sum_{j \in U(m)} p_{ik} y_j$) but, to avoid double-counting the network upstream of the central barrier *i,* this is subtracted (-z_{ik}). The habitat downstream of *m* is then subsequently found as $p_{ik}w_m$.

The entire second linear optimisation model, for maximising *the largest single undirected sub-network* is thus:

objective:

maximize Y^{max} (3.13)

subject to the following contraints:

$$y_i = \sum_{k \in O(i)} z_{ik} \qquad \forall\, i \in I \tag{3.14}$$

$$z_{ik} \le \sum_{j \in U(i)} p_{ik} y_j + p_{ik} H_i \quad \forall\, i \in I, k \in O(i) \tag{3.15}$$

$$z_{ik} \le z_{ik}^{max} x_{ik} \qquad \forall\, i \in I \tag{3.16}$$

$$w_i = \sum_{k \in O} q_{ik} \qquad \forall\, i \in I \tag{3.17}$$

$$q_{ik} \le \sum_{j \in U(m)} p_{ik} y_j - z_{ik} + p_{ik} H_m + p_{ik} w_m \quad \forall\, i \in I, k \in O(i), m \in D(i) \tag{3.18}$$

$$q_{ik} \le q_{ik}^{max} x_{ik} \qquad \forall\, i \in I \tag{3.19}$$

$$\sum_{i \in I} \sum_{k \in O(i)} c_{ik} x_{ik} \le \beta \tag{3.20}$$

$$\sum_{k \in O(i)} x_{ik} = 1 \qquad \forall\, i \in I \tag{3.21}$$

$$\sum_{i \in I} a_i = 1 \tag{3.22}$$

$$Y^{max} \le y_i + w_i - p_{ik} H_i + H_i + M^P (1 - a_i) \tag{3.23}$$

where:

y_0 = *accessible network upstream of the system sink*
I = *the set of all barriers*
i = *a single barrier in the set of all barriers*
O = *the set of options*
k = *a single option in the set of options*
$U(i)$ = *the barrier(s) immediately upstream of i*
H = *the network immediately upstream of a barrier*
j = *a single barrier in the set of upstream barriers*
y = *optimised network upstream*
z = *accessible network upstream if an option is chosen*
x = *a binary decision variable*
c = *the cost of a repair option*
β = *the total budget*
α_i = *a binary integer variable indicating whether a barrier is the parent node of the maximal subnetwork*
Y^{max} = *the network quantity associated with the maximal subnetwork*
M^P = *the largest network quantity possible (bounding variable)*
w_i = *the optimal subnetwork downstream of i*
q_{ik} = *accessible network downstream of i if option k is chosen*
$D(i)$ = *the barrier downstream of i*

The objective (3.13) is to maximise Y^{max}, the network quantity available above and below a central, undirected subnetwork barrier *i*. Constraint (3.14) and inequalities (3.15) and (3.16) are the same as the directed model and collectively define upstream permeability-weighted network. Constraint (3.17) and inequalities (3.18) and (3.19) collectively define the permeability-weighted downstream network from the central barrier *i*. Inequality (3.19) is formulated differently from the equivalent inequality (3.16) to avoid a matrix transformation. The 'set' of downstream barriers *D(i)* includes only one barrier *m*, thus assuming a dendritic network. All permeability-weighted network upstream from *m* is calculated ($\sum_{j \in U(m)} p_{ik} y_j$), subtracting the network upstream of the central barrier ($-z_{ik}$), already counted in (3.15). The permeability-weighted network downstream from *m* is then added ($+p_{ik}w_m$). Constraint (3.17) and inequality (3.18) therefore act together to calculate the permeability-weighted downstream network from barrier *i*. Inequalities

(3.20) and constraint (3.21) are the same as in the directed model. Constraint (3.22) limits the choice of subnetwork to one, as the objective is to choose a single subnetwork that is the largest possible given the budget. Inequality (3.23) defines and bounds the size of the maximal subnetwork. It is calculated as the sum of the maximal upstream y_i and downstream w_i permeability-weighted network with an adjustment to de-weight the central network segment H_i from any permeability $(-p_{ik} H_i + H_i)$. M^p is a bounding variable that is the maximum possible subnetwork, used to bound the model if no subnetwork has been selected. This model was formulated for input into the the GLPK, as a .mod file (Makhorin, 2012; see Appendix A.2).

3.5 Habitat Quality / Quantity

The incorporation of robust habitat quality or suitability models was determined outside the scope of this study, though rudimentary habitat quality modelingwas done using two treatments. Habitat Suitability Indices and habitat simulation, although necessary for generating recommendations or making decisions in this context (see Section 2.13), are complimentary but separate methods from optimisation (Wurbs & Yerramreddy, 1994). However, two treatments were created that reflected to a limited extent the relative suitability of network features to resident and migratory fish. In the first treatment, all network features were included in the habitat quantity estimate. In the second, network features representing reservoirs, river-lakes, and lakes were excluded from consideration. These treatments were chosen for two reasons: (1) important native anadromous species such as the Atlantic Salmon are known to prefer moving, oxygenated, relatively shallow water as spawning and rearing habitat (e.g., Amiro, 2006) and (2) hydroelectric dams by design create an upstream reservoir that is often very large, giving these dams a large associated habitat quantity metric and a subsequent higher probability of being selected for removal, despite these reservoirs being relatively anoxic. Optimal decision sets under these two treatments can therefore be compared to help quantify the 'reservoir effect' on relative prioritisation of barriers.

3.6 Network Quantity Estimation & Stream Width Model

The surface areas ofriver and stream features are more difficult to attain than length and less frequently available at the watershed scale (Betz et al., 2010). Given that the variability of rivers and stream widths is lost when network feature length is used as the sole quantity measure, surface area measures are a more representative and desirablequantity measure than length. However, despite network length data being available where area is not, the length data cannot easily be used complementarily in the same analysis - the two measures are not, without a conversion or intermediary metric, comparable. The cumulative contribution of small streams to overall systemic connectivity and ecosystem health is unknown but cannot reasonably be considered insignificant, as they are numerous and often contain well-oxygenated, shallow water. A further challenge is that small, inexpensive barriers such as culverts are more frequently found on small streams than large, expensive barriers. If network quantity of small stream segments is not accounted for, then the cumulative effects of small barriers will therefore not be accurately assessed in any subsequent prioritisation.

To address the lack of surface area data, relationships between stream width and a number of variables were explored and a single relationship exploited to calibrate a rudimentary stream width model. This relationship is related to the known relationship between upstream drainage area and stream width ($R^2 = 0.65$; Betz et al., 2010). The model presented here utilises total upstream network length and is less computationally intensive than calculating total upstream drainage area. The stream widths estimated using this model were used to estimate the surface area of stream segments and these areas were used to quantify river network when polygonal features were unavailable. Network quantity was summed for the network above each barrier until the headwaters of the system. The FIPEX toolset was used to do this in an iterative fashion. The model was calibrated using known widths sourced from NSPI field surveys and the NSHN GIS layers. A weak relationship was found (Pearson's R = 0.423) between total 'distance to headwaters' and stream width, with some caveats. ('distance to headwaters' <= 50 km and widths <= 27 m). The following section describes in details the process taken.

The relationships of a number of variables to stream width were explored using linear regression. In total, five metrics were obtained or derived using customized network analysis algorithms or existing information: 'Distance to Headwaters','Distanceto Mouth','Strahler' stream order, 'Shreve' stream order, and gradient.'Distance to Mouth' refers to the total network lengthbetween the centroid of a network line segment and the single outflow or sink of the system. The 'Distance to Headwaters' metric refers to the aggregate total network length between the centroid of a network line segment and the various headwaters of the system, similar to the 'upstream cell count' calculated by Betz et al. (2010). Both metrics were measured following network flow; 'distance to mouth' includes only one path to the ocean, whereas 'distance to source' includes the total length for the network between a point and all network sources.'Strahler' and 'Shreve' stream order (Strahler 1957; Shreve 1966) were obtained from the attribute table of the NSHN

(GNS, 2007; GeoNova, 2012) and were originally added to this dataset by Fisheries and Oceans Canada, Maritimes Region, using the RivEx suite of software. In the Strahler method, when two segments meet (i.e., a confluence of two streams) and each have the same stream order, the immediately downstream segment is given an order one greater than the two previous (Strahler, 1957). In the Shreve method, the segment downstream of any confluence is given an order equal to the sum of the orders meeting at the confluence (Shreve, 1966). Gradient was obtained from the original river line layer (‘wa_line’) and measures the average gradient of river line segments derived from the elevation of the start and end points of the line segment (GNS, 2007; GeoNova, 2012).

Exploratory analysis using linear regression revealed little to no apparent relationship between Strahler stream order, Shreve stream order, gradient, or 'Distance to Mouth' metrics and stream width. This was determined using scatter-plots with lines-of-best-fit and *Pearson's R* statistic. No in-depth analysis such as principal components analysis or analysis of variance was done- more robust statistical exploratory analysis may yet reveal a relationship. Upon consultation with NSPI (D. Thompson, personal communication, May 22, 2012) stream-width measurements from the NSPI dataset were revealed to have been obtained during flow measurement surveys at sites located below flow-control structures. This suggested that these widths are more variable and influenced more by regulated flow regimes than by natural systemic processes (prompting the effort to sample the 160 stream and river widths from the water polygons layer).

The stream width model used a regression on data derived from ‘Distance to Headwaters’ network analyses and calibrated using observed stream widths from site surveys and existing geospatial layers. A weak positive linear relationship between 'Distance to Headwaters' and stream width was observed. In addition, as 'Distance to Headwaters' increased, the variability of the width measures increased. 'Distance to Headwaters' was calculated for all stream segments requiring width estimates to assess where in the network the majority of them were; if stream segments in need of width estimates were close to sources of the network then the variance of known widths could be considered less important. Only 472 of 6436 (7.3% by count, 5.2% by length) line segments requiring width estimates had 'Distance to Headwaters' greater than 25 km and only 261 of 6436 line segments (4.05% by count, 2.83% by total length) had more than 50 km of network length between their centre and the source.It was therefore deemed justifiable to eliminate all width measures from the NSPI dataset if the 'Distance to Headwaters' was over 25 km, eliminating all but 32 points.The same was done with the 160 width estimates made by comparing lines to polygons in ArcGIS, leaving 17 points. Using these remaining points, a weak linear relationship was observed (*Pearson's R* = 0.423, n=49; Figure 19).

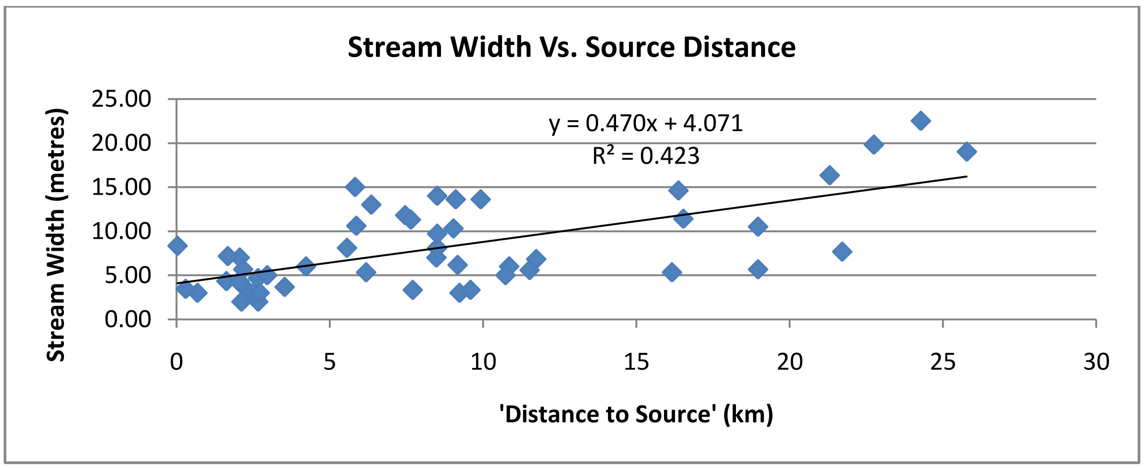

Figure 19: A relationship between 'distance to source', the total aggregate river network length between each stream segment and the system headwaters, and stream width was found for distance of 25 km or less.

The following regression equation was used to estimate stream widths for the 6436 network lines:

$$y = 0.470x + 4.071 \qquad (3.24)$$

where:

$y = stream\ width\ in\ metres$
$x = distance\ to\ source\ in\ kilometres$

After width estimates were made, areas of stream segments for the line features were made by multiplying estimated width by line length.

For the three river networks, the widths of 6436 out of 10854 line segments (59.3% by count, 59.6% by length) were estimated using the stream width model. Estimated stream widths were visually validated using known stream widths (water polygons of type 'river', feature code='WARV40'; Government of NS 2007) for river line segments with <50 km 'Distance to Headwaters' (roughly corresponding to <=27 m in width (Figures 22-25) and deemed acceptable for the purposes of this study. For river segments with > 50 km 'Distance to Headwaters', width estimates quickly became too large (Figures 26 & 27). A 'cut-off' maximum stream width value of 27 m was chosen where any stream width estimates of > 27 m were bounded to 27 m. Given that a 27 m width estimate roughly corresponded to 50 km total 'Distance to Headwaters', the number of ‘cut-off’ values was limited to 261 of 6436 line segments (4.05% by count, 2.83% by total length).

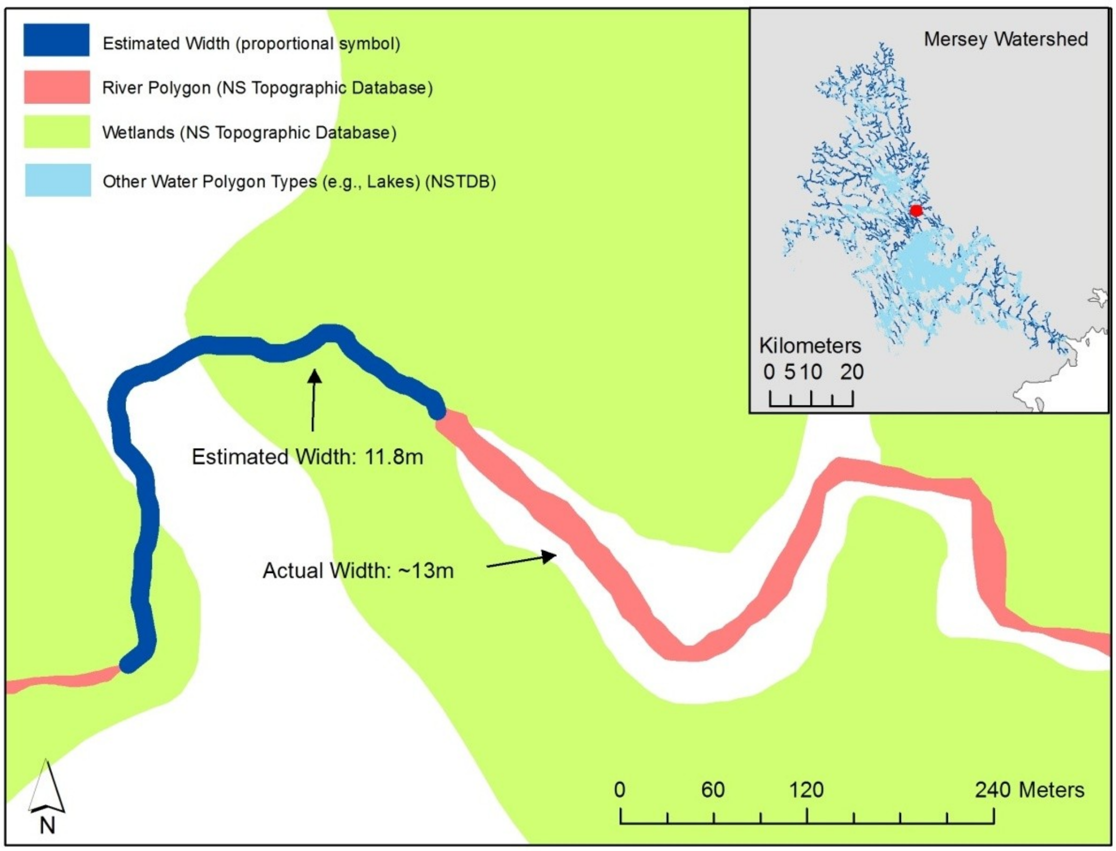

Figure 20: Comparison of model-estimated width versus actual width at a 'distance to source' of 16 km showed a difference between the predicted width and the actual width 1.2 m.

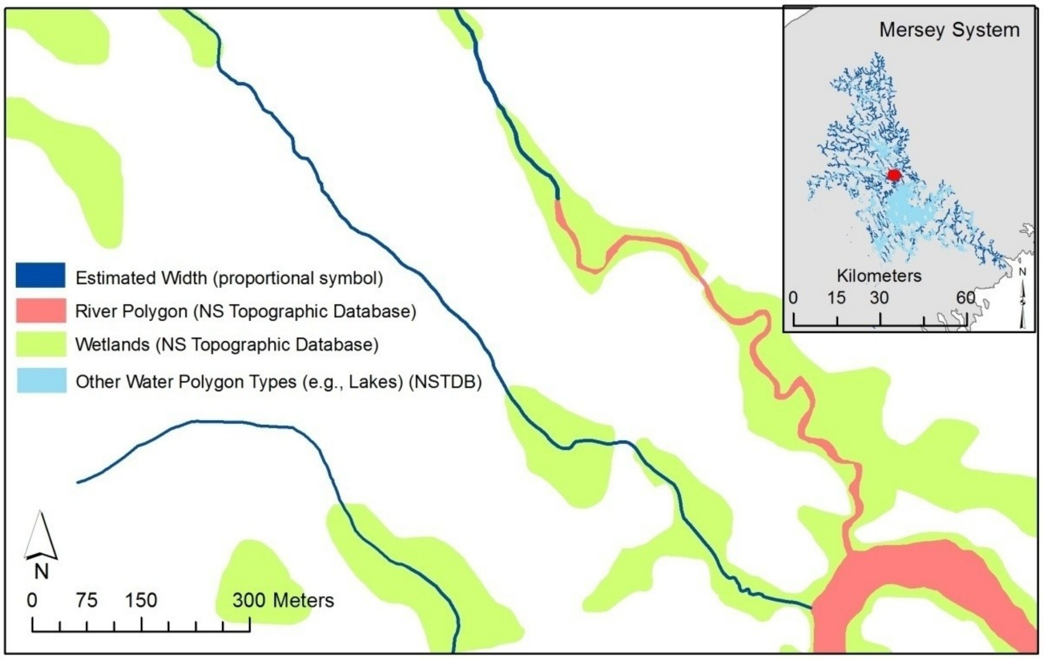

Figure 21: Comparison of model-estimated width versus actual width at a 'distance to source' of 4.5 km showed that predicted width and actual width were generally within one metre of each other.

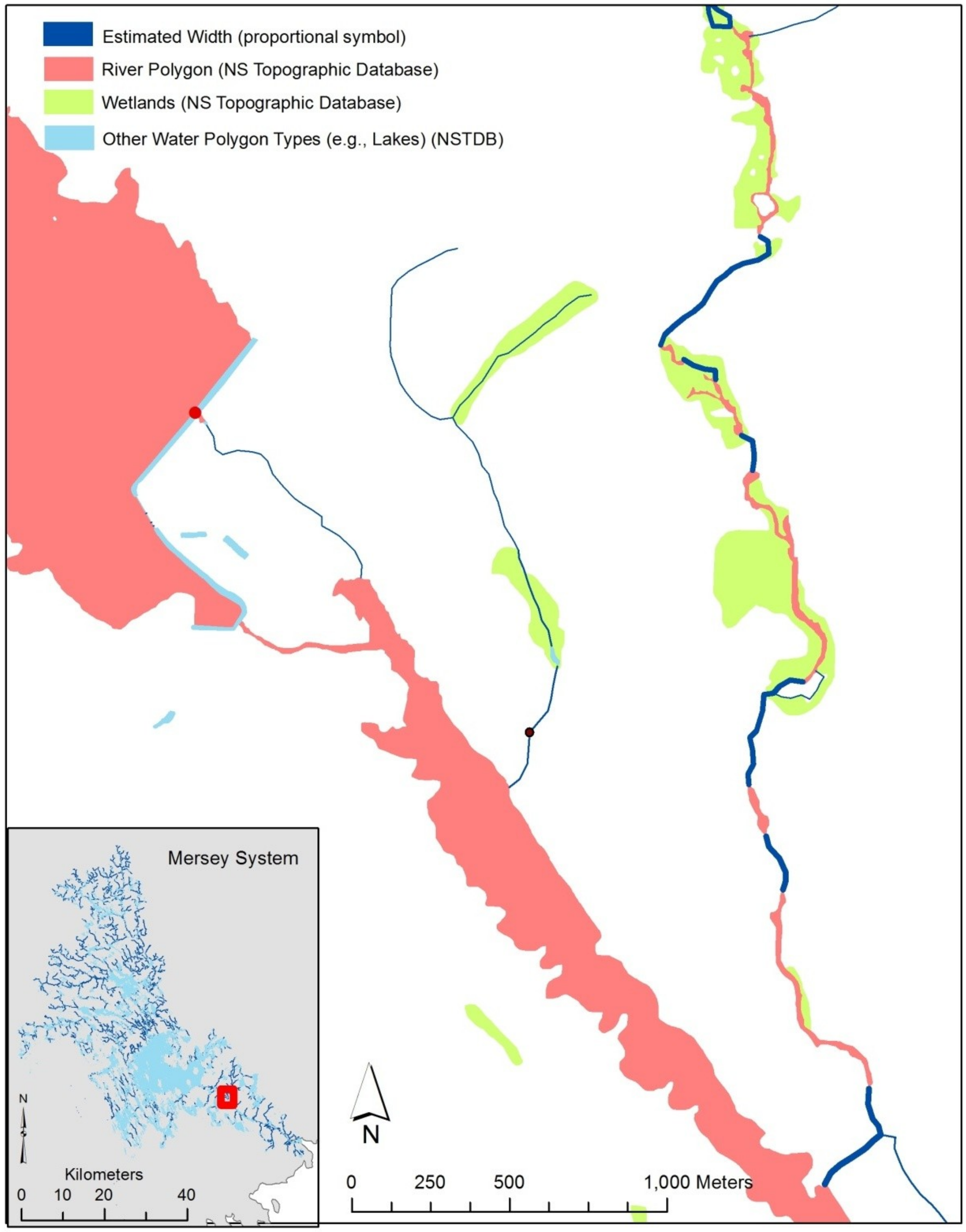

Figure 22: Comparison of model-estimated width versus actual width at a 'distance to source' of ~29 km showed model-estimated width and actual width within one metre of each other.

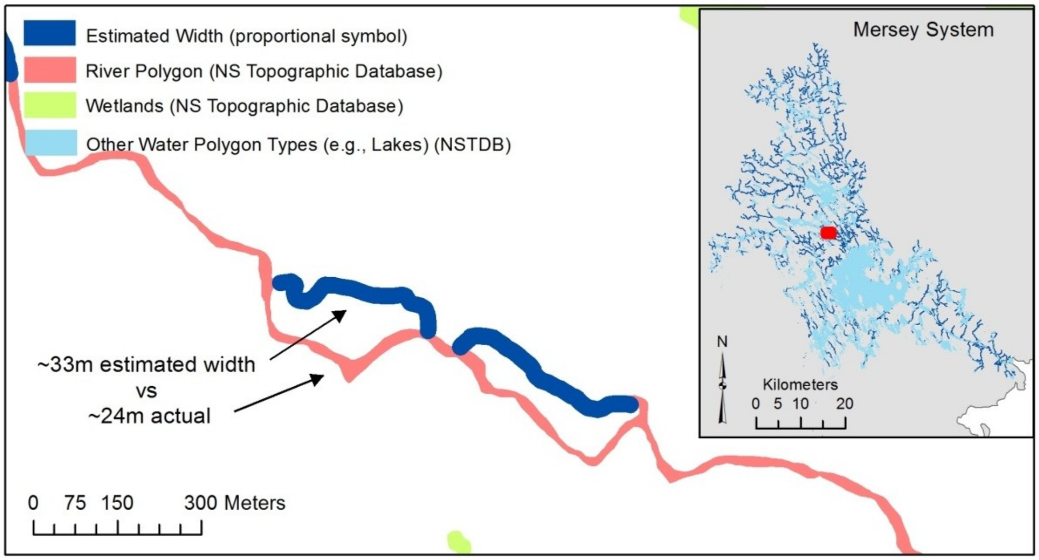

Figure 23: Comparison of model-estimated width versus actual width at a 'distance to source' of 56 km showed that model estimates become less reliable as 'distance to source' increases.

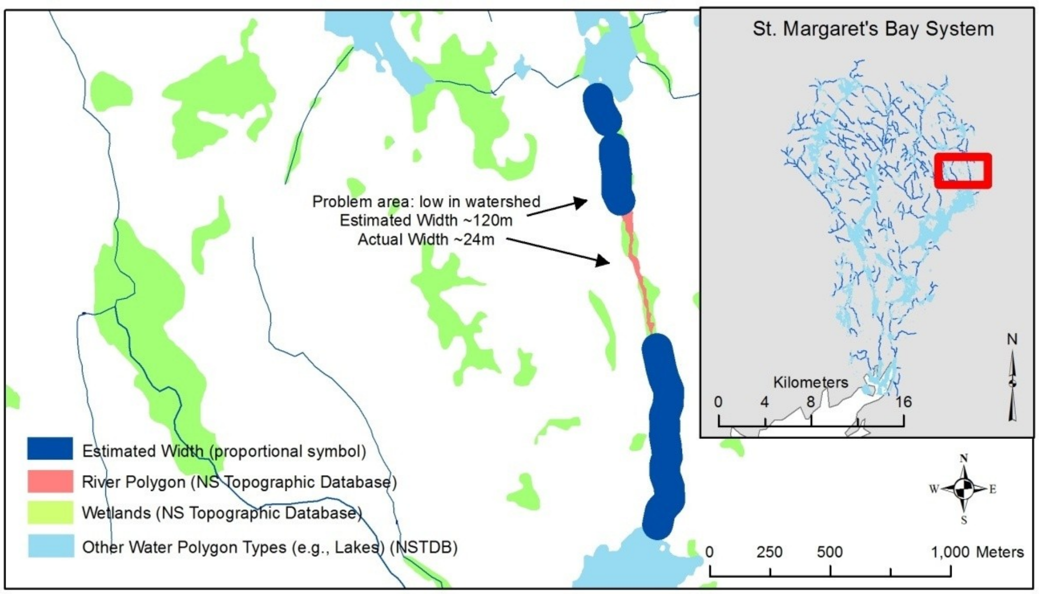

Figure 24: Comparison of model-estimated width versus actual width at a 'distance to source' of 262 km showed that model-estimated widths are unreliable at this distance.

3.7 Integration of Optimisation Models with an SDSS

The Fish Passage Extension (Fisheries and Oceans Canada, 2010) was modified to incorporate the directed and undirected optimisation models described in Section 3.4. The following additional tools and functionswere integrated into FIPEX using the Visual Basic .NET programming language and the ArcGIS ArcObjectsAPI:

- A tool to calculate 'Distance to Headwaters' and 'Distance to Mouth' metrics for lines or points.
- Functions for creating 'Connectivity' 'Habitat', and restoration 'Options' tables
- A subroutine for (1) writing / overwriting the optimisation model (with file paths to tables) and parameters file (with budget amount, sink node, time limit, etc.), (2) recursively call the GLPK solver and Gurobi solver (using Gurobi's .NET API), (3) read the output files from GLPK and Gurobi (Gurobi Optimization, Inc., 2012) and create geodatabase tables with results.
- A tool to select and help visualize barrier decision option locations.

The scripts were programmed in the Visual Basic .NET programming language and used the 'ArcObjects' API and library (the 'engine' running ArcGIS). Three tables were generated by a custom network search to be used as inputs in the optimisation models: a 'habitat' table, an 'options' table, and a 'connectivity' table.The 'habitat' table contained the barrier ID and the associated upstream network between the barrier and any upstream barriers or the headwaters of the system. The 'options' table contained a list of barrier ID's with associated project costs and the projected permeability associated with the barrier after mitigation. The 'connectivity' table was a matrix of barrier pairs -- each barrier with its immediate downstream barrier, respectively. These tables were exported by the script to the comma-separated-value format, which can be read by GLPK.The GLPK solver was used to generate an LP model file that can in turn be read by the Gurobi solver.

3.8 Analyses

3.8.1 Overview

First, customisation of an existing SDSS (FIPEX extension within ArcMap) was required to analyze river networks and prepare tables for use as inputs to optimisation analyses. Several subroutines were created to call upon the optimisation solvers from within the SDSS (described in Section 3.7), and further scripts were necessary to read the results back into the SDSS. Second, preliminary analyses were performed to assess the tractability of the models. Third, the networks were analyzed to assess the initial connectivity conditions and were reported as DCI_p, DCI_d, initial directed habitat available ($ZMAX_d$), and the size of the largest undirected subnetwork ($ZMAX_u$). Fourth, optimisation analyses for a series of budgets were conducted and the results recorded and displayed as connectivity gains versus budget curves. Fifth, a sensitivity analysis of permeability of culverts was conducted and results recorded. Sixth, a rudimentary cumulative effects assessment was done comparing expected systemic connectivity improvements if all culverts versus all dams were repaired. Seventh, a basic cumulative effects assessment was done by examining the results for indications of non-additive cumulative and interactive effects among barrier restorations. The following sections detail the methods used in each of these steps.

All analyses were done on the same PC computer with the following relevant specifications:

- Intel i5 2500k Processor
- Windows 7 64 bit
- 12Gb of DDR3 RAM
- CPU cooled to operating temperature of <60C

3.8.2 Model Tractibility

The tractability of the models was tested using all three river systems. Both the directed and undirected models were tested using the 'Area No Stillwater' treatment (hectares; one decimal place; excluding lakes and reservoir) as the network quantity measure. The models were run at budget increments of 1,000,000 CAD from 1,000,000 CAD to 10,000,000 CAD. The 32-bit Gurobi software was used rather than the 64-bit version due to a restriction imposed by the 32-bit ArcGIS software. The nodes, iterations, solve-time, and percent gap (if unsolved) were recorded.

3.8.3 Initial Connectivity Assessment

The initial states of the network with respect to longitudinal connectivity were assessed. The analyses were conducted within ArcMap (ESRI, 2012a) using the FIPEX toolset and additional customised scripts and modifications. The DCI_p and DCI_d of the systems were calculated (Cote et al.,2009) using the ‘Area’, ‘Area no Stillwater’, ‘Length’, and ‘Length No Stillwater’ network quantification methods. Network area precision was reported to the nearest 100 m^2 and network length was calculated and reported to the nearest 10 m. The initial permeability-weighted habitat accessible from the ocean was calculated using the directed optimisation model using the GLPK LP solver with the budget parameter set to zero followed by the initial largest permeability-weighted subnetwork using the undirected model and the same software and budget. The number of barriers in each network was counted and noted by type. The habitat immediately upstream of each barrier until the next upstream barrier(s) or network headwaters was calculated using the ‘Area’, ‘Area No Stillwater’, ‘Length’, and ‘Length No Stillwater’ methods. Any barriers estimated as having zero network upstream under any quantification methods were investigated for errors.

3.8.4 Optimisation

The three river systems were analyzed to find optimal decision sets for a suite of budget amounts between 15,000 CAD and 10,000,000 CAD at increments of 60,000 CAD (166 budgets). A maximum solve time of 1000 seconds (16 min 40 sec) was allotted per budget amount and a MIP Gap tolerance of 2% was allowed (the time limit was chosen to balance acceptable MIP Gap in unsolved problems with total solve time available for all analyses). Optimisation analyses were performed for each of the four network quantification treatments (i.e., ‘Area’, ‘Area No Stillwater’, ‘Length’, ‘Length No Stillwater’) and both the directed and undirected models for a total of eight budget-series optimisation analyses per system. The following results were recorded for each budget amount and treatment: (1) the decisions chosen by the optimisation solver for each budget amount, (2) whether an optimal solution was found(to less than or equal 2% MIP Gap), (3) how much time the solver took (up to 1000 seconds), (4) the size of the maximal objective subnetwork (permeability-weighted network accessible to ocean in directed model case, maximal permeability-weighted subnetwork in the undirected case), and (5) central barrier ID (for the undirected model). Connectivity gains versus budget were graphed to compare the effect of network quantification method to the shape of the curves and the relative rankings of systems, as well as to generally compare the curves to the results of previous studies.

3.8.5 Meta-analysis of Results: Culverts versus Dams

Meta-analyses of optimisation output were conducted to explore the effects of barrier mitigation by type (i.e., ‘culvert’ versus ‘dam’). The intention was to answer whether sets of culverts were prioritised over affordable dams, whether the appearance of dams in optimal decision sets corresponded with jumps in connectivity gains per dollar of budget spent, and whether results displayed non-nestedness. Furthermore, the relative gains to systemic connectivity between the mitigation of all culverts versus all dams were calculated by simulating the removal of these barriers as groups. Analyses were conducted for all three systems using results from both the directed and undirected models, using the ‘Area No Stillwater’ quantification method.

First, total impounded upstream network for each barrier was calculated and aggregated by barrier type using each quantification method. Second, to analyse the effects on longitudinal connectivity of each type of barrier as a group, longitudinal connectivity gains relative to the initial state of each of the three networks were measured upon separate simulated removals of all culverts and all dams.Connectivity was quantified by calculating the DCI_p,DCI_d (Cote et al., 2009), and $ZMAX_u$, both before and after simulated barrier mitigation. Network connectivity gains were compared between the culverts and dams using the percent gain in DCI values and the gain in $ZMAX_u$ relative to the size of the river system (maximum ZMAX). The $ZMAX_d$ was omitted from measurement because, upon scaling relative to the size of the river system, the result yields the DCI_d and therefore would provide the same general result. The difference between each of the three metrics before and after barrier removal was calculated for each network quantity treatment and each river system. The difference in DCI between treatments was calculated as:

$$\Delta DCI = (DCI^{dams} - DCI^{Initial}) - (DCI^{culverts} - DCI^{Initial}) \tag{3.25}$$

Where $DCI^{culverts}$ is defined as the DCI after culverts are removed, and DCI^{dams} is the DCI after dams are removed.The $ZMAX_u$ gain was scaled to the maximum $ZMAX_u$ possible in the network (i.e., the total size of the network) before the gains under the two treatments were compared to enable comparisons across watersheds, thus:

$$\Delta ZMAX_u = (ZMAX_u^{culverts} - ZMAX_u^{init})/\, ZMAX^{sys} \tag{3.26}$$

The results from the directed and undirected models and the 'Area No Stillwater' treatment were used to create graphs and figures to aid in the interpretation of results. The number of projects in the optimal decision sets at each incremental budget amount and the presence of dams in the output, for all dams, were included. All occasions where dams were affordable but did not appear in the final decision set were recorded. Connectivity gains versus budget curves were graphed also using the DCI as the measure of connectivity. The number of barriers in the optimal decision sets, the connectivity gains versus budget, the connectivity gains per dollar versus budget, and nestedness of dams in optimal repair results were graphed together to give a qualitative overview.

Optimisation results were explored to find examples of occasions where sets of culverts were selected over affordable dams. These were highlighted and mapped. Cases of non-nestedness were also highlighted and mapped.

3.8.5 Culvert Permeability Sensitivity Analysis

Given their numbers and unpredictable and variable conditions, the permeability of culverts is difficult, relative to dams, to assess on a watershed scale. An objective of this study was to address the challenge of quantifying culvert permeability by asking whether accurate estimation of this variable is necessary to be confident in restoration priorities attained through optimisation, the goal being to expand knowledge on the value of 'perfect information' with regard to culvert permeability. To test the sensitivity of optimised priorities to the permeability of culverts, a basic sensitivity analysis was done. To achieve this, the permeability of culverts was randomised between 0 and 1 (to two decimal places) and optimisation analyses run for two budget amounts for each system and the directed and undirected models. The 'best guess' optimal decision sets were found using the culvert permeability estimation of 0.5 (bi-directional). A series of 30 optimisations was conducted using repeated randomisation of the permeability parameter. Maximal network size was recorded under simulated repair of the barrier sets found by optimisations using both the randomised permeabilities and the best-guess permeabilities of 0.5. For each randomisation the random permeabilities were assumed to be one possible scenario of 'perfect information' and, given that network size is a function of permeability, the size of the resulting network was re-calculated using these random permeabilities.

Two budgets for each system and model type were chosen for the sensitivity analysis. These budgets were selected based on the slope of the habitat gain per dollar versus budget curve and by the number of culverts fixed at each budget for each system and optimisation model. Budget amounts were avoided both where very few or no culverts were likely to appear andwhere close to or all barriers were likely to appear in the optimal decision sets. The former was avoided to ensure a number of culverts actually appeared in the output. The latter was avoided because as the number of barriers appearing in the optimal decision sets approaches the total number of barriers on the network then, logically, the difference between 'best-guess' decision sets and those attained using randomised permeabilities approaches zero. A hypothesized'sensitive' budget was selected from a steeply-sloped region and an 'insensitive' budget selected from a relatively flat region of the connectivity gains per dollar versus budget curves. Ideally, all budgets would have been assessed for sensitivity to permeability; however, current limitations of computer software and hardware made this excessively time-consuming. A total of 30 randomisations and associated optimisation runs were done for each budget, each model type, and each system (treatment total = 12). The measure of network quantity used in analyses was the 'Area No Stillwater' treatment with a maximum allowed solve-time of 1000 seconds. After the 30 randomisations for each treatment, if the optimisations remained unsolved (i.e., greater than 2% MIP Gap), up to ten optimisation runs out of 30 were discarded; 20 randomisations was deemed the minimum necessary to provide a meaningful average of maximal network size under random permeability treatments. This was done to avoid underestimating the difference between the optimal maximal network assuming random permeabilitiesand the sub-optimal maximal network found using ‘best-guess’ repair decisions (the underestimation would be up to the % MIP Gap remaining in the unsolved model run).

The sensitivity analysis quantified the value of ‘perfect information’ and opportunity cost of acting upon imperfect information (i.e., culvert permeability = 0.5) using two measures: (1) foregone connectivity gain, and (2) budget wasted.Both (1) and (2) were expressed in absolute terms and percent relative to a baseline. The foregone connectivity gain was found by first calculating the connectivity gain relative to the initial state of the system when repairing the best-guess barriers. This value was then subtracted from the connectivity gain attained using the optimal decision set found under the randomised permeability treatment. The mean of the differences for all randomisations for a given treatment was calculated and was then expressed as a percentage of absolute total connectivity gain in hectares as compared to the initial state of the system. The budget wasted was calculated by attempting to answer the question: *approximately how much budget would have been required to achieve the connectivity gain attained by the best-guess decisions (as estimated using the randomised permeabilities)?* This metric was calculated by dividing the connectivity gains attained using the optimal decision set found using randomised permeabilities, *ZMAX*, by the budget, B_i, to get the connectivity gains per dollar. This efficiency measure was then used to calculate the hypothetical cost, C_h, of optimally attaining the connectivity gains that were sub-optimally attained using the best-guess decision set, $ZMAX_{bg}$:

$$C_h = {ZMAX_{bg}} \Big/ \left({ZMAX_r} / {B_i} \right) \tag{3.27}$$

This was then subtracted from the initial budget, B_i, yielding the budget wasted, B_w.

$$B_w = B_i - C_h \tag{3.28}$$

This method assumes that the connectivity gains versus budget curves are linear and constant, which is not accurate; however, this approach was deemed adequate to yield some insight into the magnitude of sacrifice made through acting upon imperfect information.

3.8.6 Cumulative Effects Analysis: Controlling for Cost

Two key factors were theorized to contribute to observations of slopes >1 in connectivity gains versus budget curves, slopes >0 in connectivity gains per dollar versus budget curves, and sudden jumps in both cost and synergistic cumulative effects due to spatial interdependence. Thus, cost was controlled in separate analyses of all three systems and both types of connectivity which, in the absence of other unrealized factors, were believed to isolate the relative contribution of both cost and synergism to observed irregular patterns in these curves. The expected results and hypothesized evidence of synergism were outlined (Figure 25 &Figure 26) with synergism defined as (1) increasing marginal gains observed in the connectivity gains versus budget curves and (2) positive slope observed in the connectivity gains per dollar versus budget curves (see Diefenderfer et al., (2012) for a similar approach).

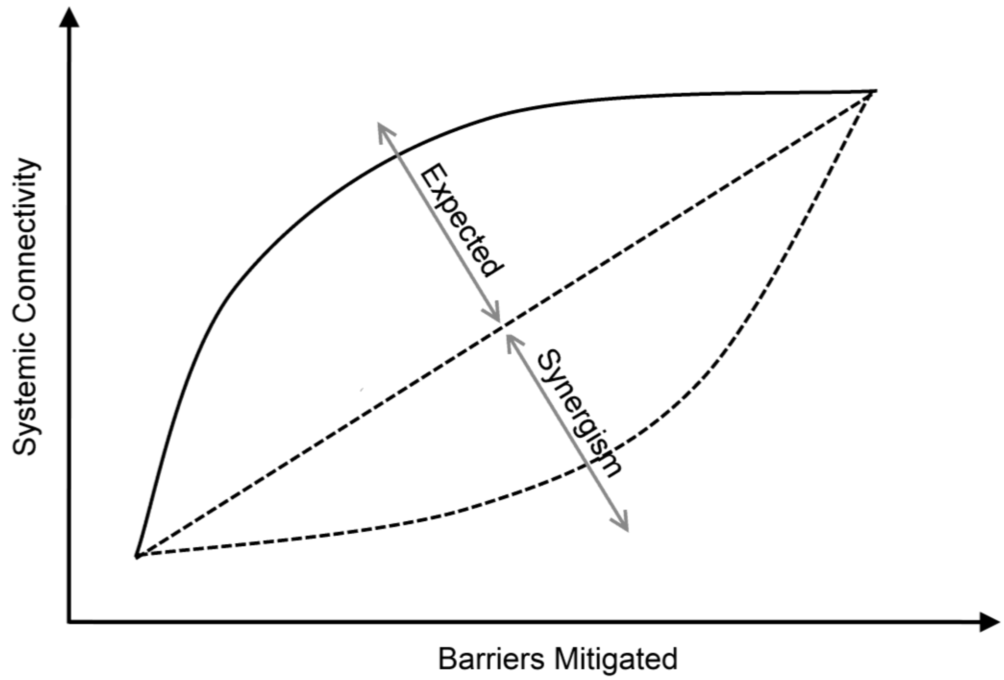

Figure 25: Expected results of connectivity versus budget based on past studies (e.g., O'Hanley & Tomberlin, 2005; O'Hanley, 2011) and hypothesized evidence of synergistic cumulative effects of barrier mitigation due to spatial interdependence.

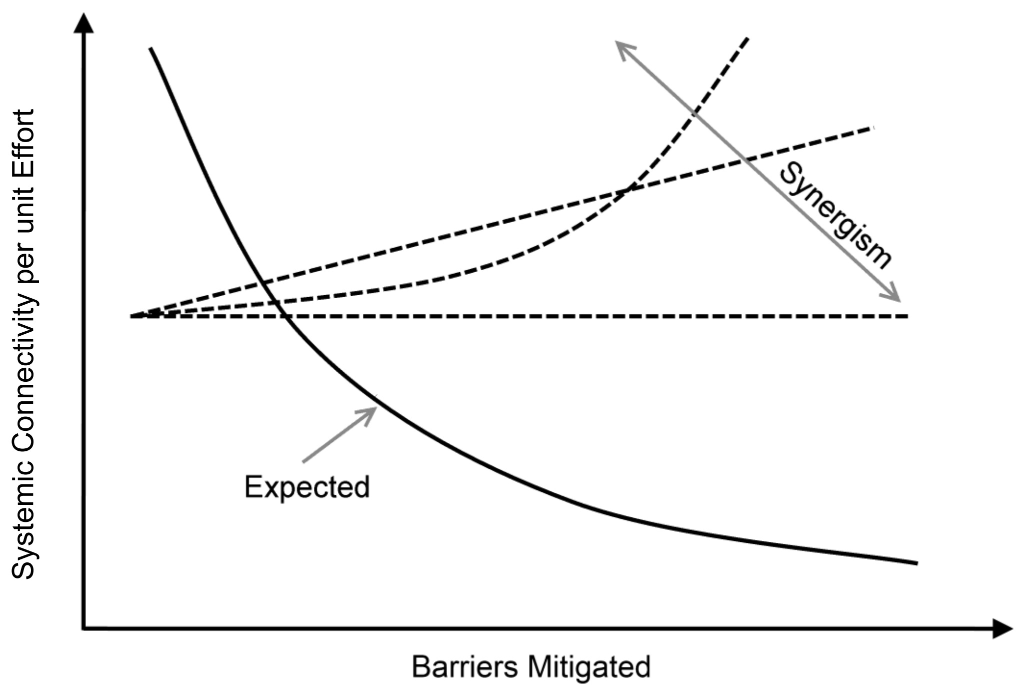

Figure 26: Expected results of versus budget and hypothesized evidence of cumulative effects of barrier mitigation due to spatial interdependence.

All barrier repair costs were set equal and optimisations conducted, incrementally increasing the number of total allowable decisions in the optimal decision output. Sudden jumps or non-negative connectivity gains per dollar versus budget slopes could be attributabed to the delay of the inclusion of efficient projects in optimal decision sets caused by high project cost were thus removed. Analyses were conducted using the 'Area no Stillwater' quantification method and a maximum allowable solve-time of 500 seconds for each optimisation analysis at increments until all dams were repaired. Analyses were conducted for the undirected and directed model on the three river networks. Results were investigated to observe nestedness of dams (the large number of culverts presented a challenge to investigate for non-nestedness, without further customized script development, deemed outside the scope of this research), in output decision sets. Connectivity gains versus total number of barriers fixed and connectivity gains per dollar versus total number of barriers fixed were graphed. Basic linear regressions were used to find lines of best fit for the resulting datasets.

CHAPTER 4: RESULTS

4.1 Model Tractibility

The computational burden of the directed model was relatively low compared to that of the undirected model. Both the GLPK and Gurobi solvers were able to solve all directed model optimisations, but the GLPK solver often could not solve the undirected models in the time allotted. As a result, only Gurobi was used for reported analyses. Mersey, the largest system in terms of barriers and options was the most challenging network to analyse (Table 4: Solve-time Reported for the Directed Model and 'Area No Stillwater' Treatment.**).** The undirected model required much longer in general to solve (Table 5), though it solved successfully for all budget amounts. The maximum solve time for the undirected model at any budget was approximately 2 hr 50 min.

Table 4: Solve-time Reported for the Directed Model and 'Area No Stillwater' Treatment.

SHEET HARBOUR

Budget ($000s)	Time (sec)
1000	0.06
2000	0.02
3000	0.02
4000	0.02
5000	0
6000	0
7000	0
8000	0
9000	0
10000	0

MERSEY SYSTEM

Budget ($000s)	Time (sec)
1000	0.1
2000	0.1
3000	0.1
4000	0.1
5000	0.1
6000	0.2
7000	0.1
8000	0.1
9000	0.1
10000	0.1

ST. MARGARET'S BAY

Budget ($000s)	Time (sec)
1000	0
2000	0.1
3000	0.1
4000	0.13
5000	0.1
6000	0.1
7000	0
8000	0
9000	0
10000	0

Table 5: Solve-time Reported for the Undirected Model and 'Area No Stillwater' Treatment.

SHEET HARBOUR

Budget ($000s)	Time (sec)
1000	605.6
2000	10,229.7
3000	2105.5
4000	421.9
5000	44.1
6000	108.2
7000	11.2
8000	30.2
9000	31.1
10000	31.1

MERSEY

Budget ($000s)	Time (sec)
1000	128.26
2000	183.09
3000	634.9
4000	235.9
5000	1,056.6
6000	350.69
7000	756.3
8000	819.35
9000	7.31
10000	0.8

ST. MARGARET'S BAY

Budget ($000s)	Time (sec)
1000	730.49
2000	212.5
3000	27.56
4000	36.1
5000	50.58
6000	17.6
7000	25.21
8000	17.14
9000	10.67
10000	9.59

4.2 Initial Connectivity Assessment Results

The states of longitudinal connectivity of the Mersey, St. Margaret's Bay, and Sheet Harbour systems were assessed by calculating the DCI_p, DCI_d, the accessible permeability-weighted network from the ocean (directed subnetwork; $ZMAX_d$), the largest undirected subnetwork ($ZMAX_u$), and the total habitat available under four treatments of network connectivity quantification (Table 6). The $DCI_d$and $ZMAX_d$ were lower than the DCI_p and $ZMAX_u$ in all three systems using all four quantification methods with the Mersey system showing the greatest gap. The Mersey system was significantly more affected in connectivity to and from the ocean, whereas the connectivity within the system was higher than the others.

Table 6: Initial connectivity (DCI_d, DCI_p), Directed Subnetwork ($ZMAX_d$), Largest Undirected Subnetwork ($ZMAX_u$), and Total Systemic Network for Three River Systems Studied.

Watershed and Quantification Method	DCI_d	DCI_p	$ZMAX_d$	$ZMAX_u$	Total	Units
Mersey						
Length	1.72	46.21	25.97	933.42	1647.12	km
Length No Stillwater	1.76	45.87	20.65	625.36	1172.94	km
Area	0.52	64.37	160.33	24145.01	29449.32	ha
Area No Stillwater	4.38	27.28	91.02	786.60	2077.07	ha
St. Margaret's Bay						
Length	0.63	14.88	2.36	82.42	371.78	km
Length No Stillwater	0.94	15.39	2.28	54.73	241.23	km
Area	0.12	18.51	4.81	1158.11	3843.48	ha
Area No Stillwater	**2.14**	18.72	4.50	55.80	209.61	ha
Sheet Harbour						
Length	22.99	36.92	203.23	319.00	884.21	km
Length No Stillwater	23.75	37.62	162.31	247.67	683.11	km
Area	18.44	36.56	1066.34	2001.00	5781.34	Ha
Area No Stillwater	27.59	51.14	210.40	323.55	762.69	Ha

Connectivity assessments differed noticeably depending on the quantification method used. For example, the Mersey system's directed and undirected connectivity assessments differed particularly between the 'area' (DCI_d = 0.52; DCI_p = 64.37) and 'area no stillwater' methods (DCI_d = 4.38; DCI_p = 27.28). Variation was also observed in the undirected connectivity assessment of the Mersey system between the 'length' (DCI_p = 45.87) and 'length no stillwater' (DCI_p = 64.37) quantification methods.

4.3 Optimisation Results

From the results of the directed model, the connectivity gains observed as budgets increased differed significantly from those of the undirected model. A general trend of decreasing connectivity gains as budgets increased was interpreted from the results of the undirected model (Figure 31-Figure 34). The results of the directed model, however, were inconsistent, showing varying marginal longitudinal connectivity gains and connectivity gains per dollar as budgets increased (Figure 27-Figure 30), exhibiting dominant positive (St. Margaret's Bay), negative (Sheet Harbour) and near-linear (Mersey) trends. In addition, the results of the directed model showed pronounced jumps as compared to the results of the undirected model.

Relative rankings of the three river systems in terms of both connectivity returns and connectivity returns per dollar changed when measured in absolute terms (ZMAX; ha or km) or scaled to the total size of the system (DCI_d). Economies of scale (i.e., increasing connectivity returns per dollar as budgets increase) at certain budget amounts were observed in the results of the directed model analyses on all three systems – the Mersey system at higher budgets, the St. Margaret's Bay system at moderate to high budgets, and the Sheet Harbour system at lower budgets.

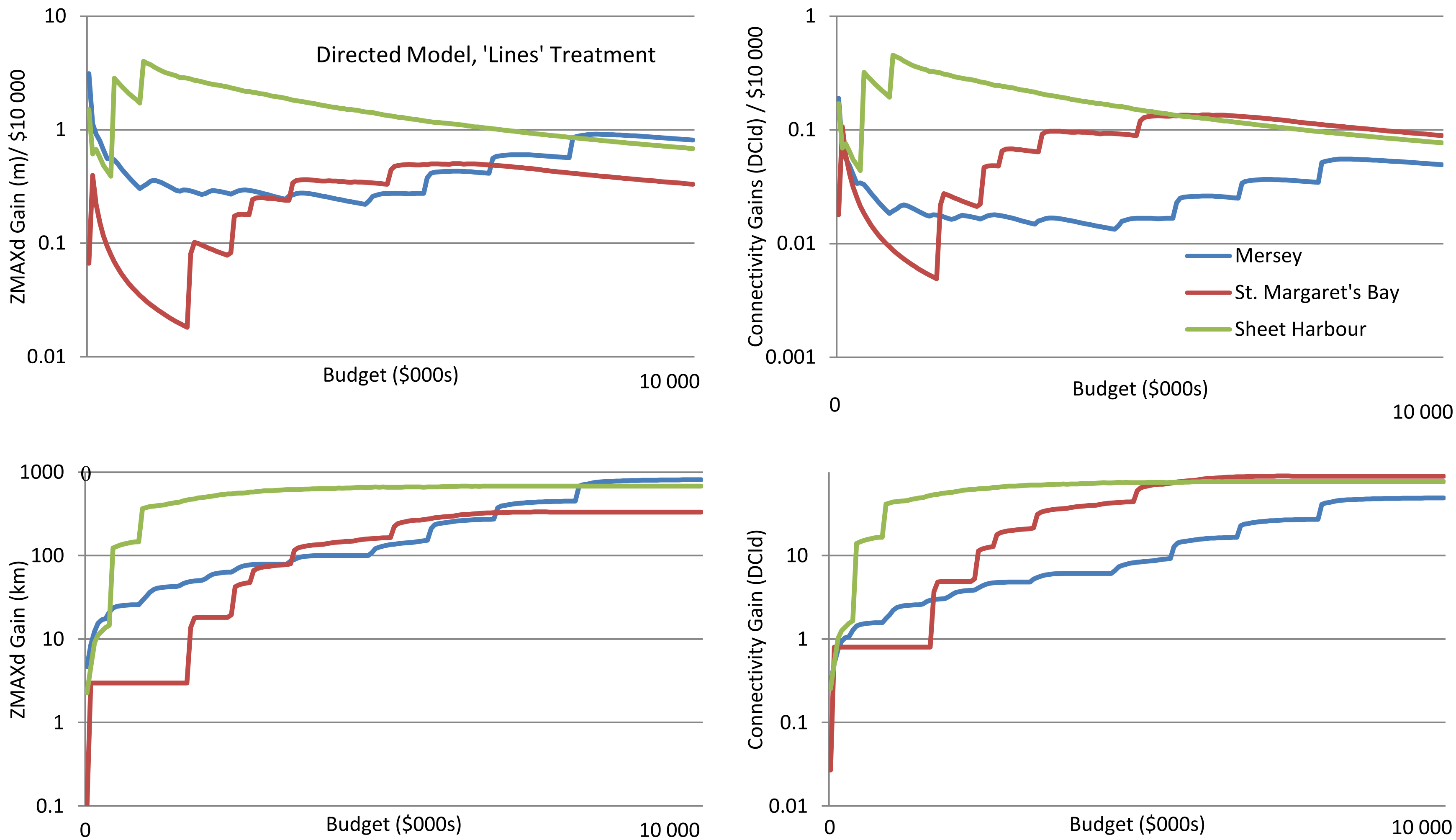

Figure 27: Optimisation results for the directed, 'Lines' treatment showing connectivity gains and connectivity gains per 10,000 CAD total budget.

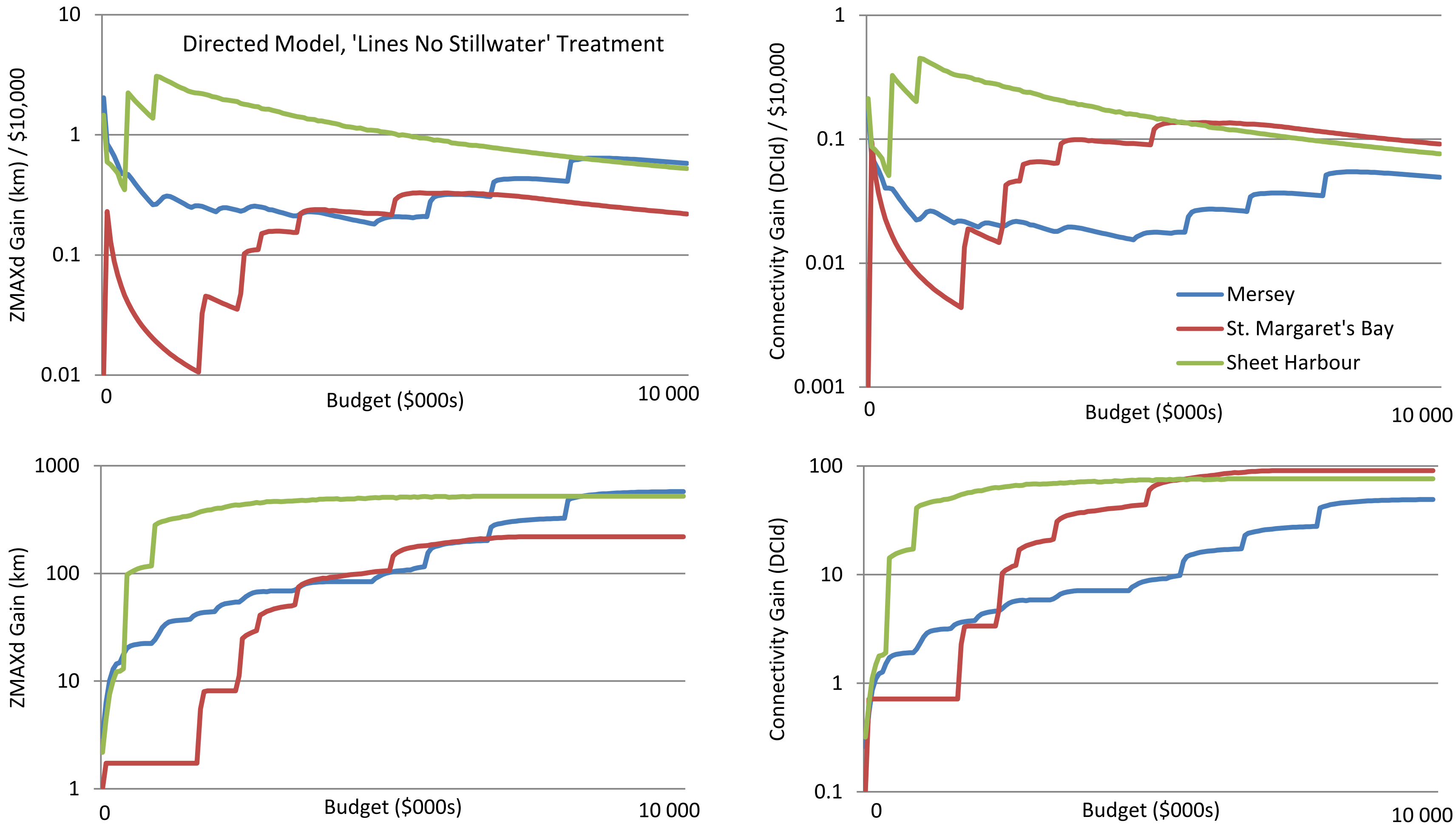

Figure 28: Optimisation results for the directed, 'Lines No Stillwater' treatment showing connectivity gains and connectivity gains per 10,000 CAD versus budget.

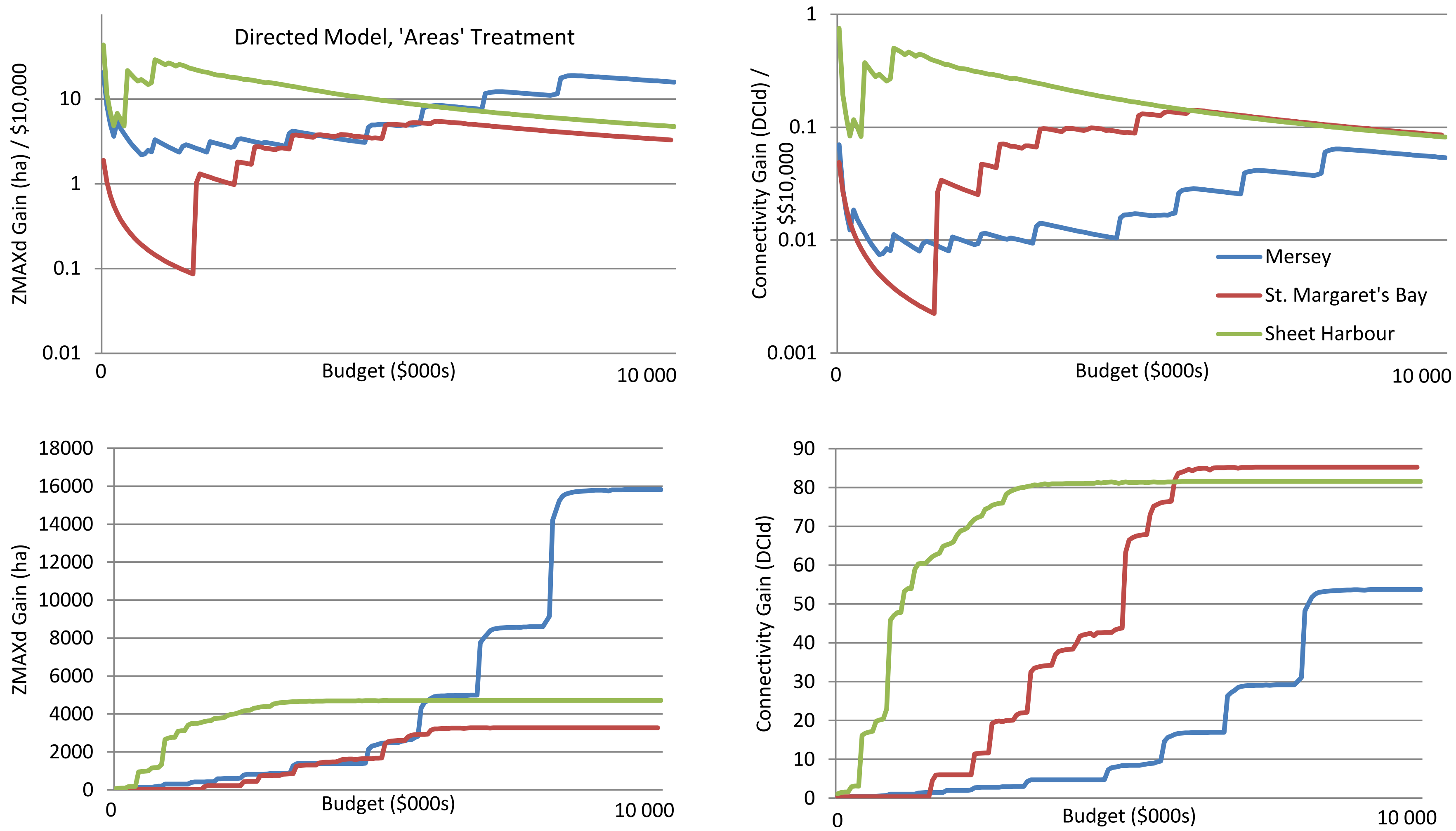

Figure 29: Optimisation results for the directed, 'Areas' treatment showing connectivity gains and connectivity gains per 10,000 CAD total budget.

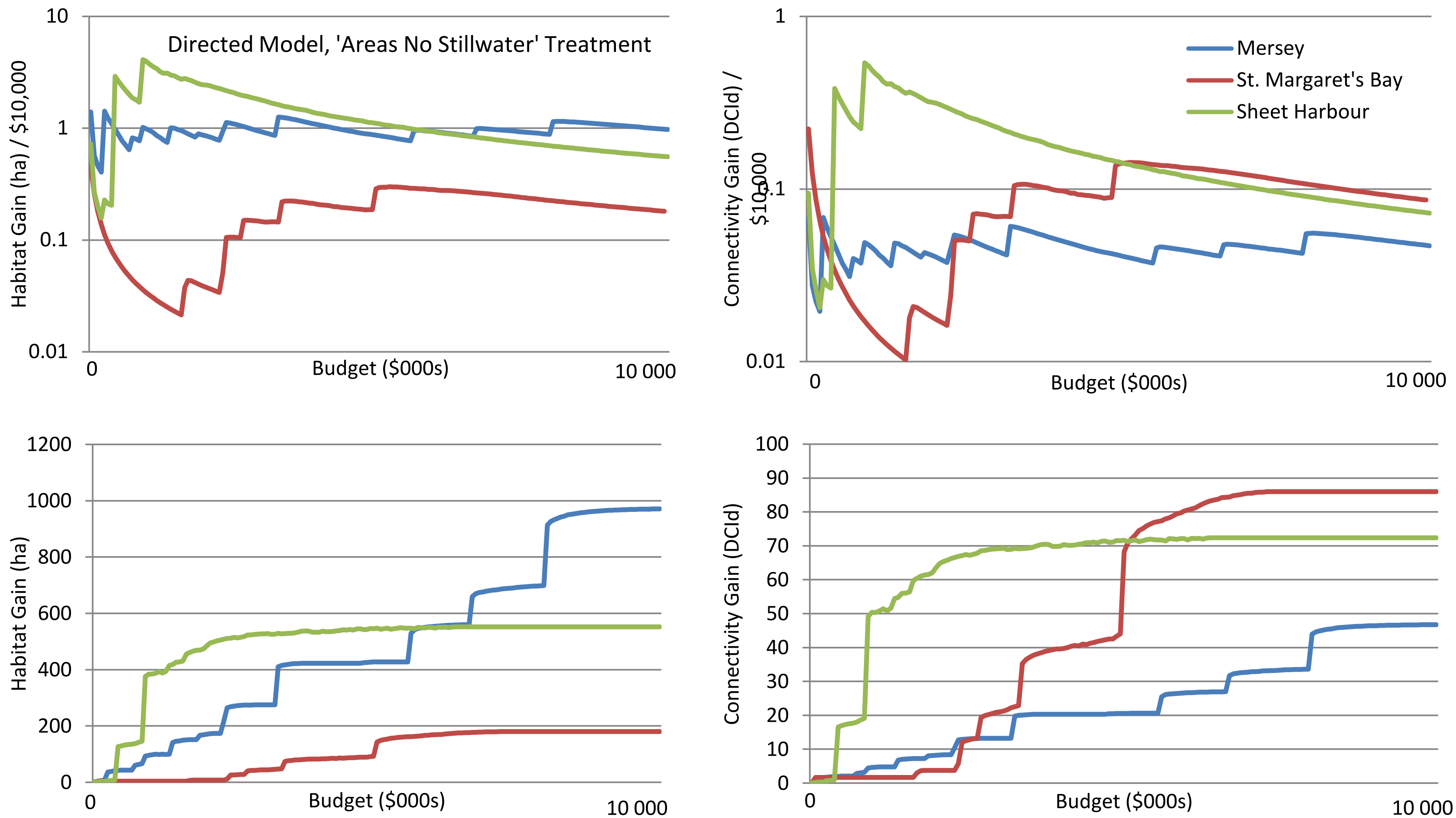

Figure 30: Optimisation results for the directed, 'Areas No Stillwater' treatment showing connectivity gains and connectivity gains per 10,000 CAD budget.

The results of the undirected optimisation model showed an exponentially declining trend of marginal gains as budgets increased, with certain exceptions. Plots of both absolute connectivity gain and gains per dollar revealed a similar trend between river systems (Figure 31-Figure 34). However, in the results of the Mersey system using the 'Area No Stillwater' quantification method, increasing marginal connectivity versus budget and connectivity returns per dollar versus budget were observed at higher budget amounts.

Relative rankings of the systems with respect to estimated connectivity gains and connectivity gains per dollar changed notably between measures of absolute connectivity gain versus gain scaled to the size of the total network (i.e., ZMAX versus DCI) for both the results of the directed and undirected models. These changes varied between the four network quantification methods. The St. Margaret's Bay system, for example, ranks lowest in terms of gains to $ZMAX_u$ but highest when $ZMAX_u$ was scaled to the size of the river system (except from results of using the 'Areas' method of network quantification, in which case it ranks second highest in terms of scaled $ZMAX_u$). In the results of the undirected model, the Mersey system consistently ranked higher than either the St. Margaret's Bay or Sheet Harbour systems in terms of connectivity gains per dollar and total connectivity gains as quantified by absolute gains, but ranked lowest in terms of gains scaled to the total size of the system. Note that the DCI_p metric was not calculated at each budget amount due to software and processing time limitations, though $ZMAX_u$ is a component of DCI_p and maximising $ZMAX_u$ also maximises DCI_p (see Section 3.4.3, Figure 3). From the results of the directed model, the Mersey system yielded greatest absolute gains at low budget amounts but was overtaken at higher budget amounts by the Sheet Harbour system. However, when scaled to the total size of the river system (i.e., DCI_d), the Mersey system is overtaken in terms of connectivity returns at higher budget amounts by the St. Margaret's Bay system.

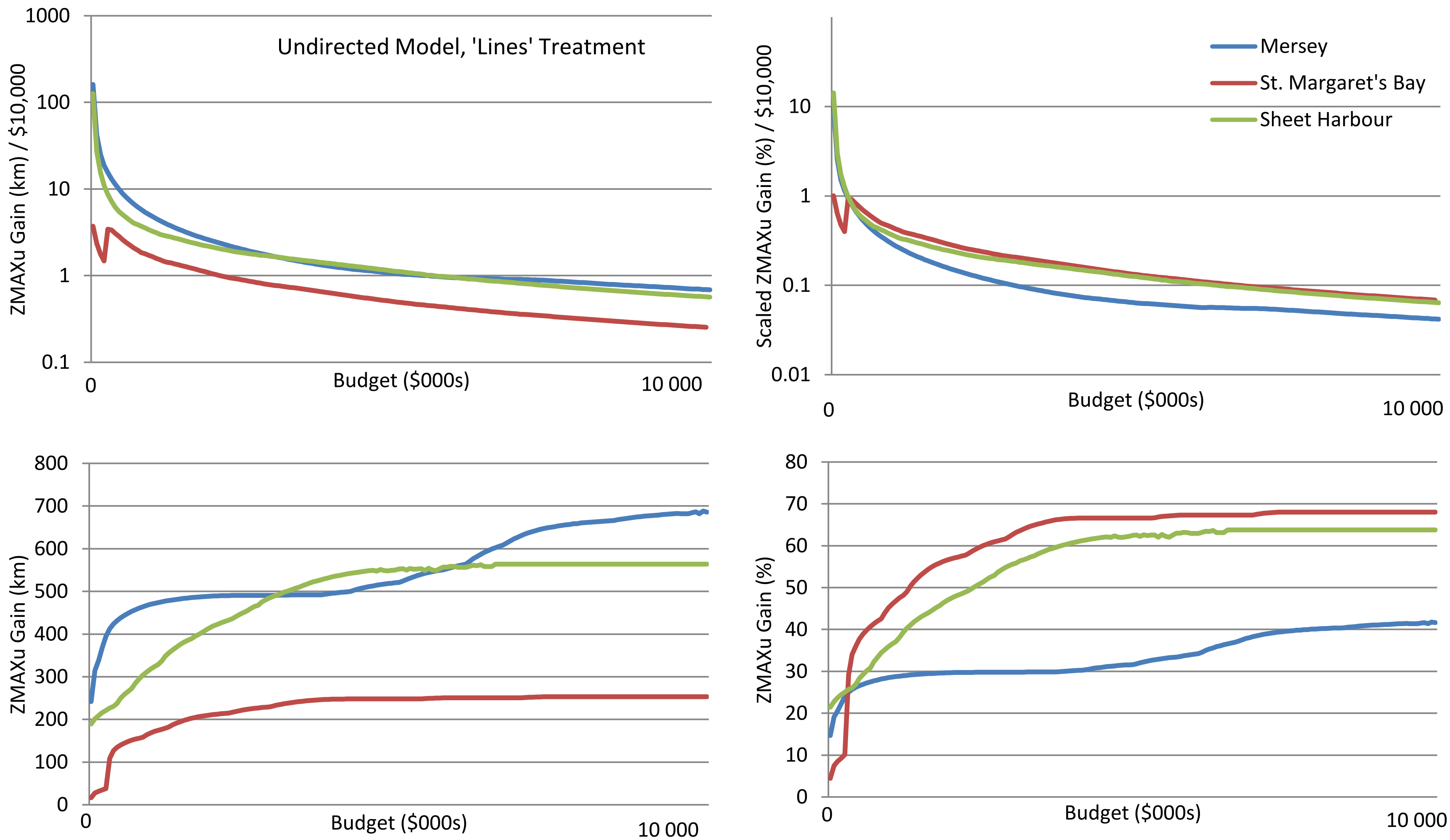

Figure 31: Optimisation results for the undirected, 'Lines' treatment showing connectivity gains and connectivity gains per 10,000 CAD total budget.

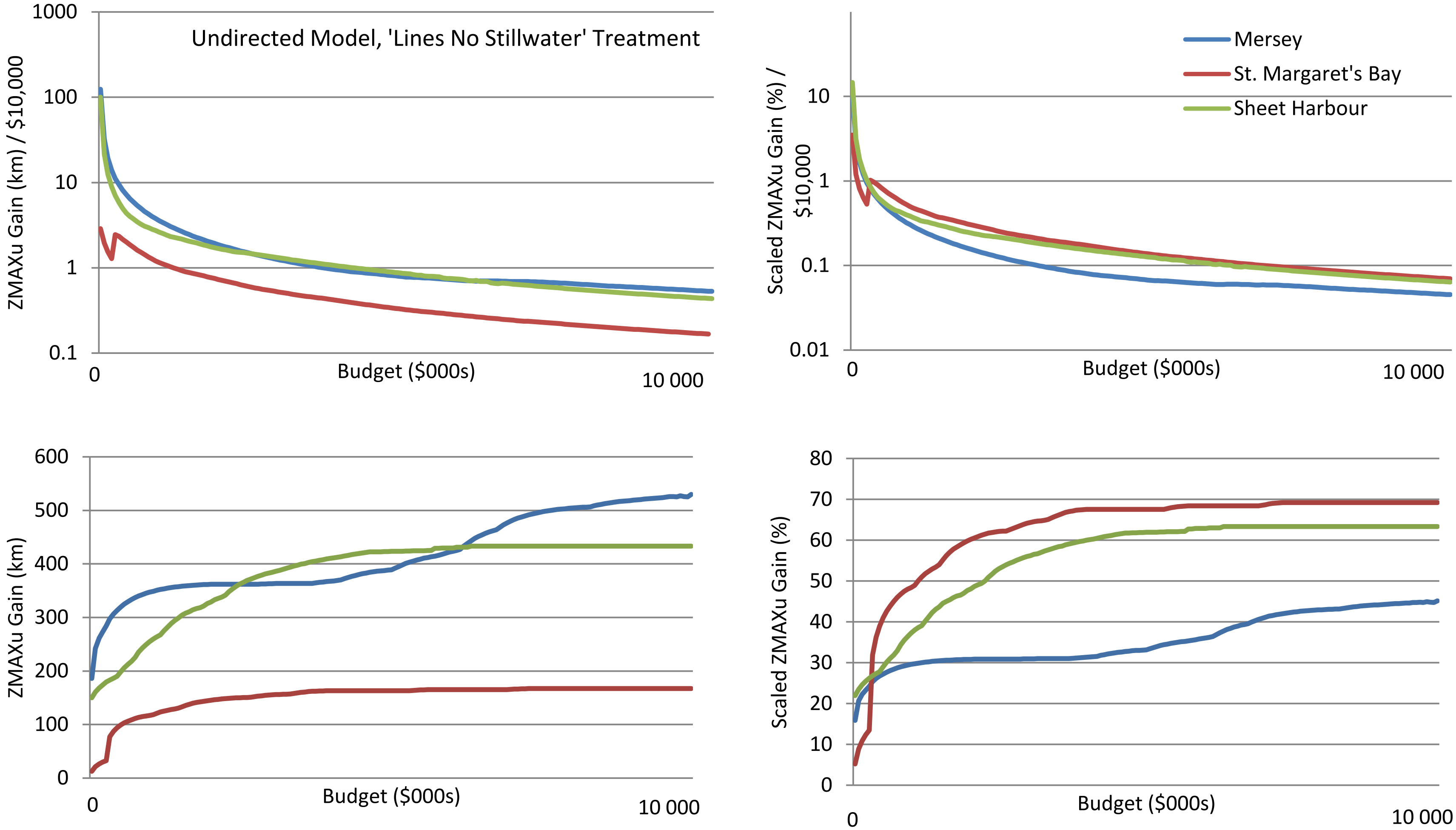

Figure 32: Optimisation results for the undirected, 'Lines No Stillwater' treatment showing connectivity gains and connectivity gains per 10,000 CAD versus budget.

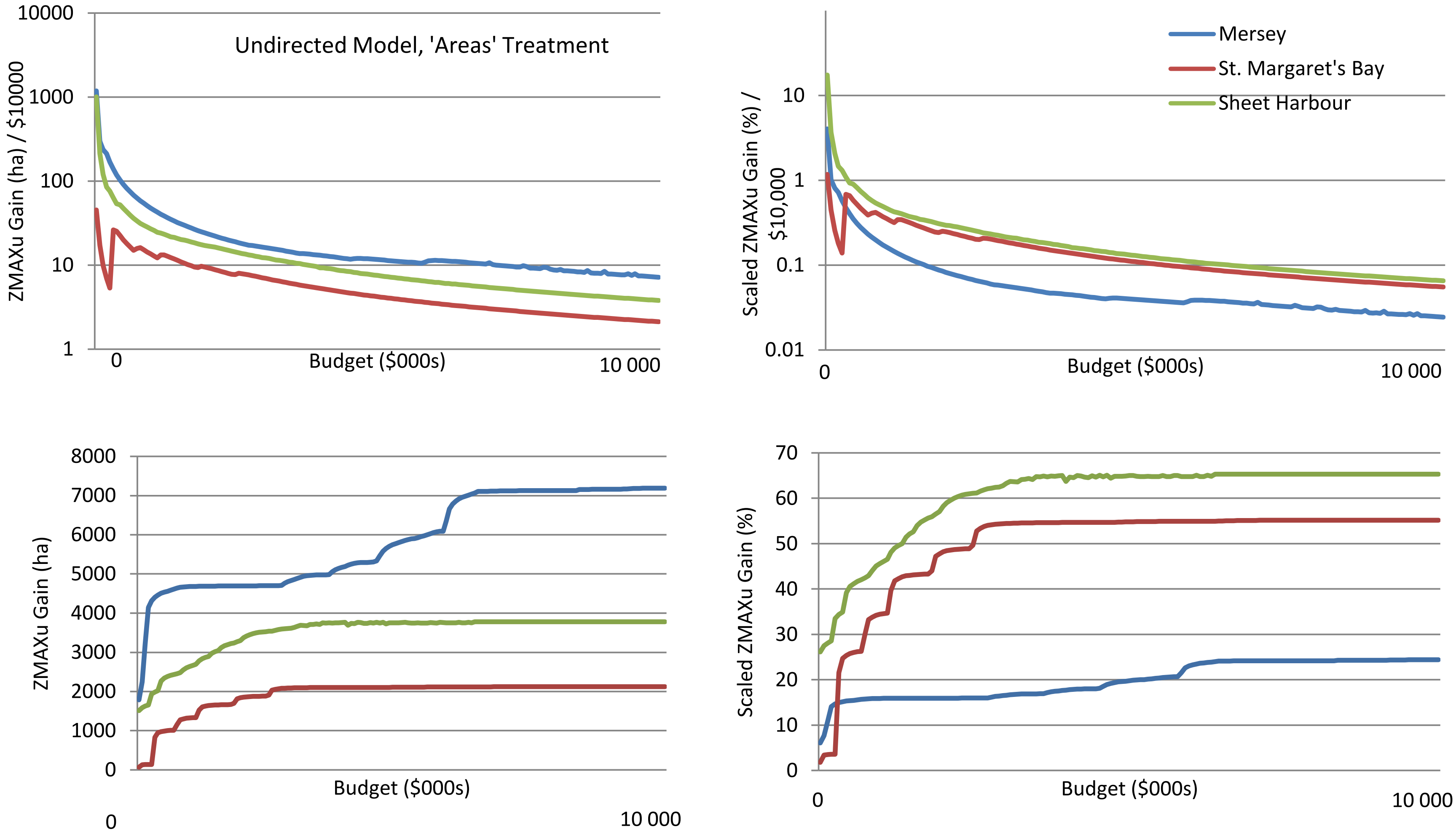

Figure 33: Optimisation results for the undirected, 'Areas' treatment showing connectivity gains and connectivity gains per 10,000 CAD budget.

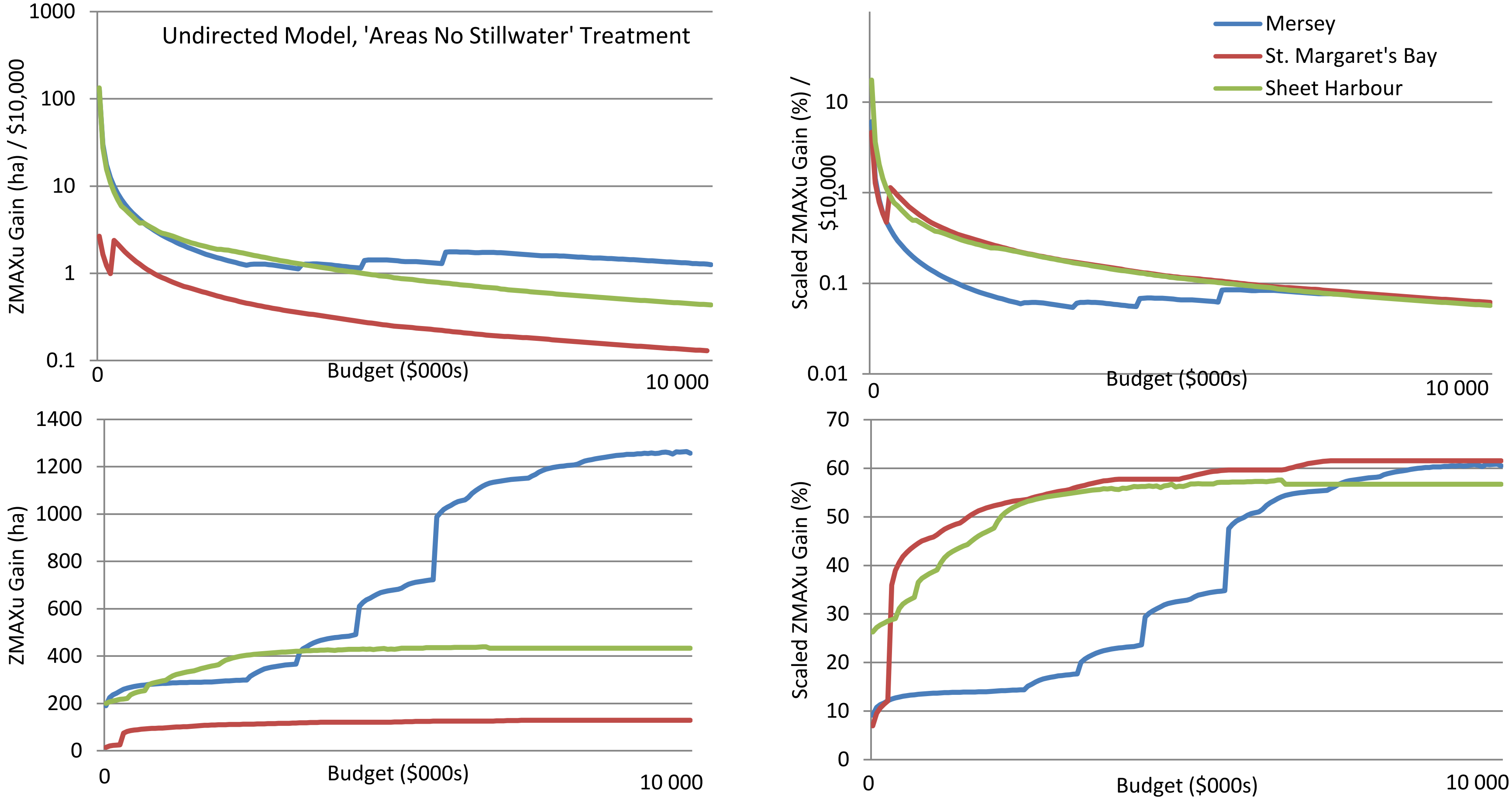

Figure 34: Optimisation results for the undirected, 'AreasNo Stillwater' treatment showing connectivity gains and connectivity gains per 10,000 CAD versus budget.

4.4 Meta-Analysis Results: Culverts versus Dams

Results of this analysis indicate that, as a group, dams have a greater impact on connectivity than culverts on these three hydroelectric systems (Table 7). However, when the relative cost of barriers was considered via optimisation analysis, culverts were found to be an occasionally higher priority than dams. Results of assessment of the impounded river network immediately upstream of barriers show that a consistently greater proportion of river networks are impounded by dams than culverts despite the large numbers of culverts **(**Figure 35**),** though the proportion of network impounded by culverts was (~30 – 45%) when stillwater was omitted. Directed connectivity gains upon simulated removal of dams, as measured by the ‘Areas’ treatment (DCI_d:71.07 – 92.92), were always much greater than those of simulated mitigation of all culverts (DCI_d: 0.01 – 1.79). This result was consistent when lentic waterbodies were excluded under the ‘Areas No Stillwater’ treatment (dam removal gains DCI_d: 63.92 – 86.25; culvert removal DCI_d: 0.01 – 1.82). Undirected connectivity gains upon simulated removal of dams were also greater than gains of simulated culvert removal for both the ‘Areas’ treatments (damremoval gains DCI_p: 23.05 – 49.81; culvert removal gains DCI_p: 3.61-7) and the ‘Areas No Stillwater’ treatment (dam removal gains DCI_p: 32.64 – 55.26; culvert removal gains DCI_p: 6.75 – 9.89). Comparison of culvert removal gains to dam removal gains (taking an average within systems of the ‘Area’ and ‘Area No Stillwater’ gains) in a paired t-test assuming equal variance supports the hypothesis that there is a significant difference in connectivity gains between dam (M=72.85,SD=7.58) and culvert removal (M=0.93,SD=1.24) for directed connectivity ($t(2)=-13.25$, p(two-tail)<0.006).For undirected connectivity, similarly strong differences were found between the DCI_pgain found in culvert removal (M=7.38,SD=0.78) and dam removal simulation (M=42.28,SD=5.45; $t(2)=-8.94$, p(two-tail)<0.012). Similar results were found for the difference in gains in the largest undirected subnetwork, $ZMAX_u$ (scaled to total network size for and averaged between the ‘Areas’ and ‘Areas No Stillwater’ treatments,) between dam removal (M=50.89, SD=0.59) and culvert removal scenarios (M=6.19,SD=3.83), with $ZMAX_u$ showing significantly greater gains upon dam removal ($t(2)=-16.32$,p(two-tailed)<0.004).

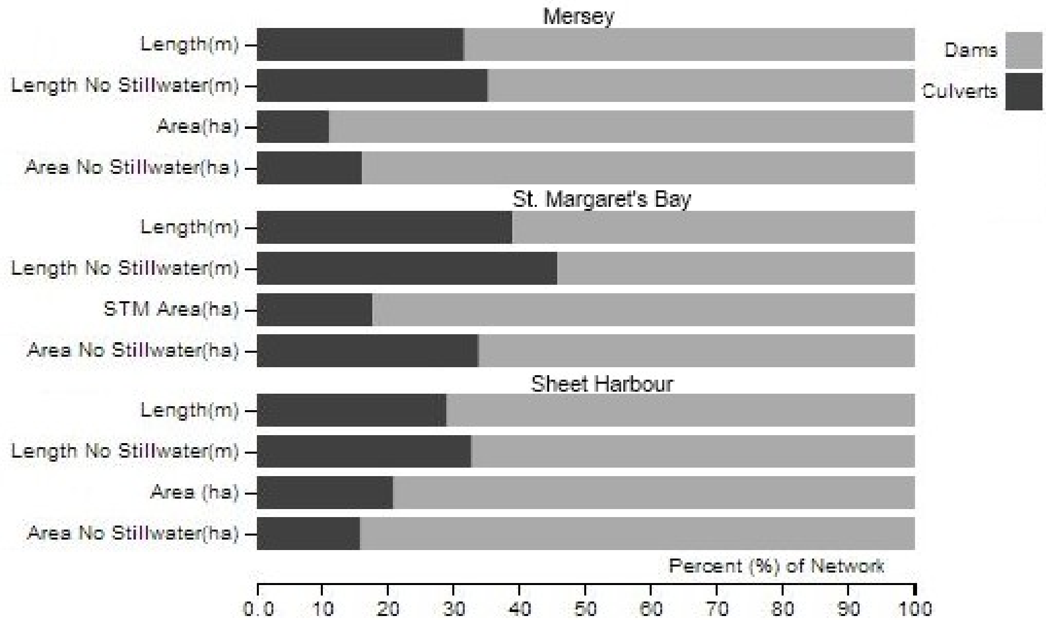

Figure 35: Aggregated measures of impounded network by four methods of quantification for each system.Results show consistently lower aggregate impounded river network by culverts than dams. Notable differences exist between aggregated impounded network by barrier type between quantification methods, with the amount of network impounded by culverts particularly reduced when area quantification measuures were used versus length.

Table 7: Results of a Basic Cumulative Effects Analysis in which all Culverts were Mitigated and, Separately, all Dams were Removed.

		DCI_d Before	DCI_d After	ΔDCI_d (dams - culverts)	**Mean ΔDCI_d (for t-test)**	DCI_p Before	DCI_p After	ΔDCI_p (dams - culverts)	**Mean ΔDCI_p (for t-test)**	$ZMAX_u$ Before (ha)	$ZMAX_u$ Before (%)	$ZMAX_u$ After (ha)	$ZMAX_u$ After (%)	$\Delta\%ZMAX_u$ (dams - culverts)	$\Delta ZMAX_u$ (dams - culverts)	$\Delta\%ZMAX_u$ Gain (dams - culverts)	**Mean $\Delta\%ZMAX_u$ (for t-test)**
Mersey																	
Area	Dam repair	0.5	93.42	92.7	**89.26**	64.37	87.42	16.05	**32.28**	24145.20	81.99	28073.60	95.33	13.34	1358.10	4.61	**11.17**
	Culvert repair		0.72				71.37					26715.50	90.72	8.73			
Area No Lakes	Dams repair	4.38	90.63	85.82		27.28	82.54	48.51		786.60	37.87	1425.13	68.61	30.74	368.43	17.74	
	Culverts repair		4.81				34.03					1056.70	50.87	13.00			
St. Margaret's Bay																	
Area	Dam repair	0.12	81.1	80.98	**78.21**	18.50	68.31	46.20	**39.39**	1158.10	30.13	3141.50	81.74	51.60	1845.70	48.02	**42.40**
	Culvert repair		0.12				22.11					1295.80	33.71	3.58			
Area No Lakes	Dam repair	2.14	77.58	75.43		18.72	61.18	32.57		55.80	26.62	162.69	77.62	51.00	77.09	36.78	
	Culvert repair		2.15				28.61					85.60	40.84	14.22			
Sheet Harbour																	
Area	Dams repair	18.44	89.51	69.28	**65.69**	36.92	80.77	37.23	**30.43**	2001.00	34.61	5155.20	89.17	54.56	3058.90	52.91	**46.97**
	Culvert repair		20.23				43.54					2096.30	36.26	1.65			
Area No Lakes	Fixing repair	27.59	91.51	62.1		51.14	83.78	23.62		323.55	42.42	677.20	88.79	46.37	313.00	41.04	
	Fixing repair		29.41				60.16					364.20	47.75	5.33			

Examination of the results of optimisation revealed that pronounced jumps in connectivity returns per dollar versus budget and connectivity gains versus budget corresponded with dam appearance in the optimal decision set. These jumps were larger and more frequently observed in the results of the directed model (Figures 37-39). Appearance of dams in the optimal decision sets from the undirected model results rarely corresponded with a noticeable jump in connectivity gains or connectivity gains per unit effort, though the results for the Mersey system were an exception (Figures 40-42). Appearance of culverts in output decision sets corresponded to dominant trends of decreasing connectivity gains and connectivity gains per dollar, whereas the appearance of dams in optimal decision sets corresponded with pronounced jumps in connectivity gains and connectivity gains per dollar versus budget. A steep jump was observed in the undirected model results from the St. Margaret's Bay system analysis at an approximate budget of $900,000 corresponding to the remediation of an impassable channel in the centre of the system representing an inter-basin transfer between the east and west watersheds (Figure 41).

Dams almost always appeared in optimal decision sets arrived at by the directed connectivity model when they were affordable, unless other dams out-ranked them (Figures 37-39). The Five Mile Lake dam in the St. Margaret's Bay system and the Governor and Seloam Lake dams in the Sheet Harbour system were notable exceptions, with culverts out-ranking these dams until budgets higher than expected (i.e., budgets greater than the threshold affording dam repair). In contrast, optimal prioritisations arrived at by the undirected connectivity model showed that dams were frequently absent from optimal decision sets despite being affordable (Figures 40-42). Sets of culverts were found, especially in the output of the undirected model, to cumulatively outweigh the benefits of one or multiple dams at certain budgets (e.g., Figure 36). Mixed sets of culverts and dams frequently occurred in model output, with affordable dams absent from decision sets – this was observed in results of both models for all three river systems. Nestedness figures revealed that the results from all models and all systems (except for St. Margaret's Bay, undirected model) displayed, to some extent, non-nestedness of dams. That is, dams appearing at a lower budget were absent at one or more higher budgets.

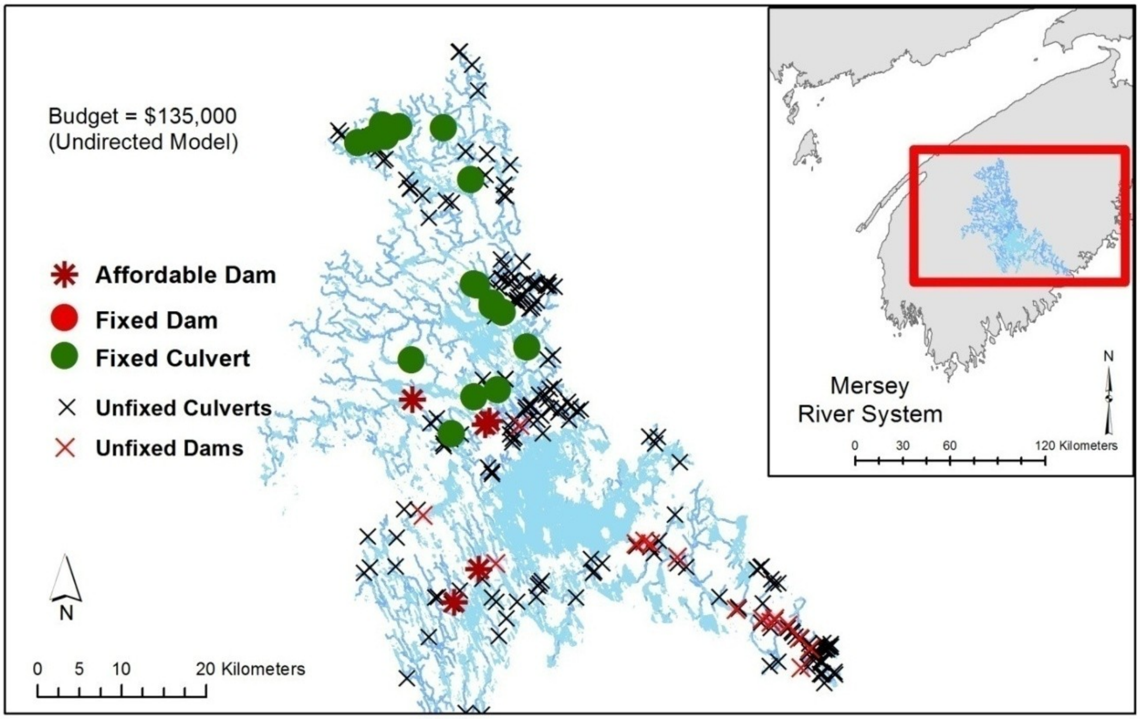

Figure 36: Map of the Mersey river system illustrating cumulative effects of culverts (green circles) in the optimal decision set outweighing the effects of affordable dams (red asterisks) at a budget of $135,000 for the undirected model.

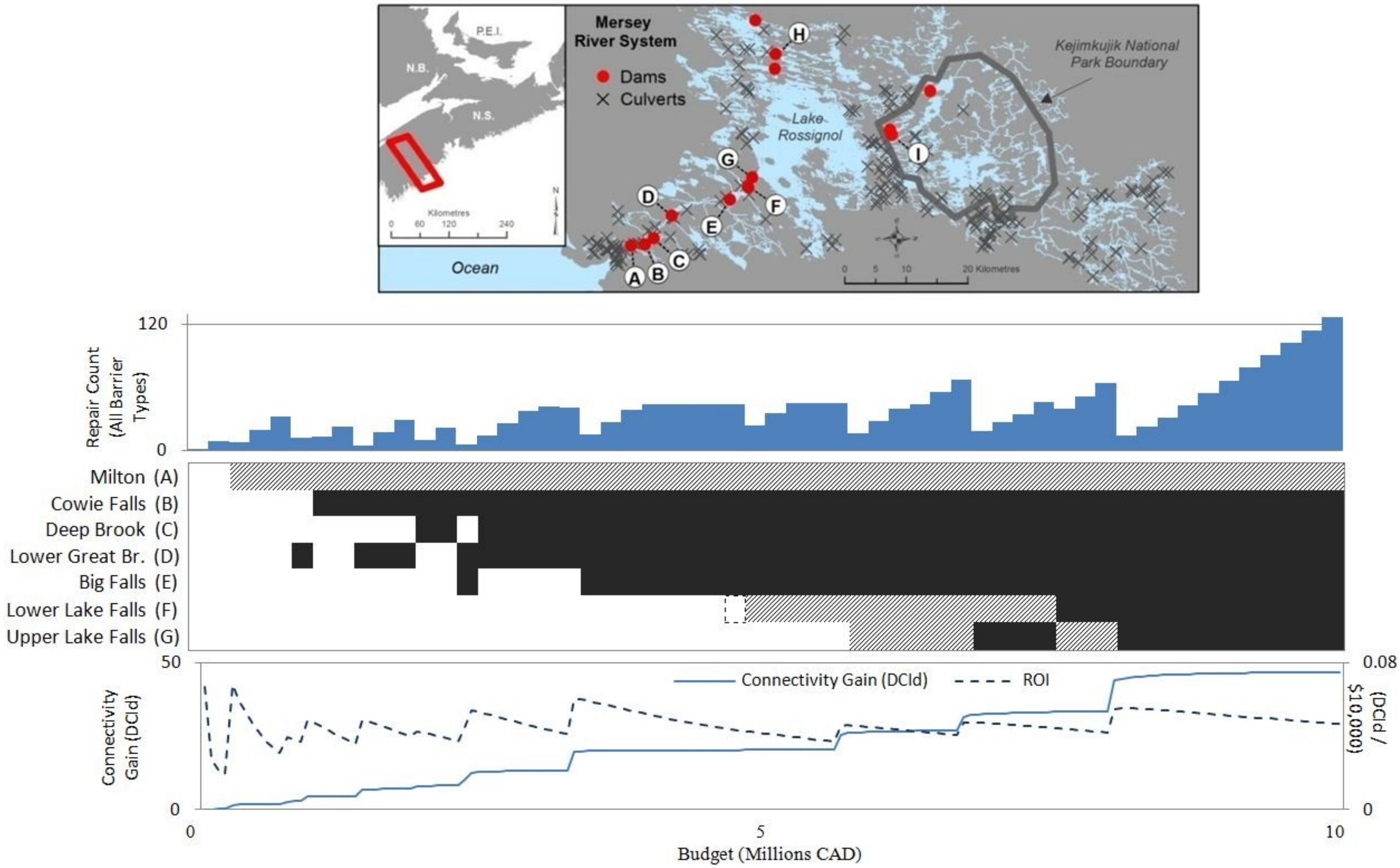

Figure 37: Results of the directed model ('Area No Stillwater' network quantification method) for the Mersey system highlighting appearance of dams. corresponding jumps in total number of barriers appearing in optimal decision sets, DCI_d gains per 10,000 CAD versus budget, and DCI_d gains versus budget. Dashed cell borders highlight occasions where dams were affordable but did not appear in optimal decision sets. Hatched cell shading indicates a 'half-repair' (i.e., upstream or downstream connectivity restoration project only).

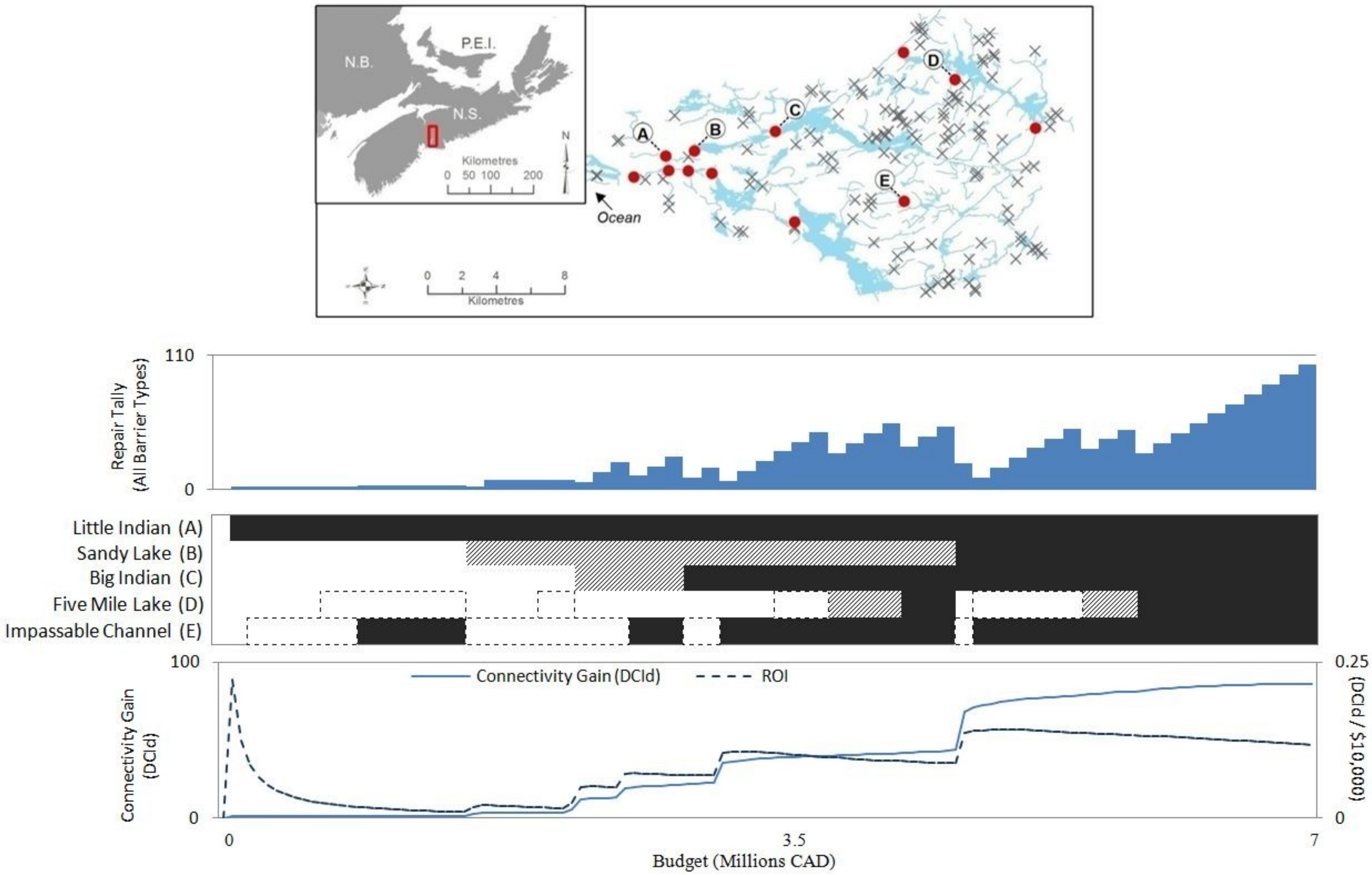

Figure 38: Results of the directed model ('Area No Stillwater' network quantification method) for the St. Margaret's Bay system highlighting appearance of dams. corresponding jumps in total number of barriers appearing in optimal decision sets, DCI_d gains per 10,000 CAD versus budget, and DCI_d gains versus budget. Dashed cell borders highlight occasions where dams were affordable but did not appear in optimal decision sets. Hatched cell shading indicates a 'half-repair' (i.e., upstream or downstream connectivity restoration project only).

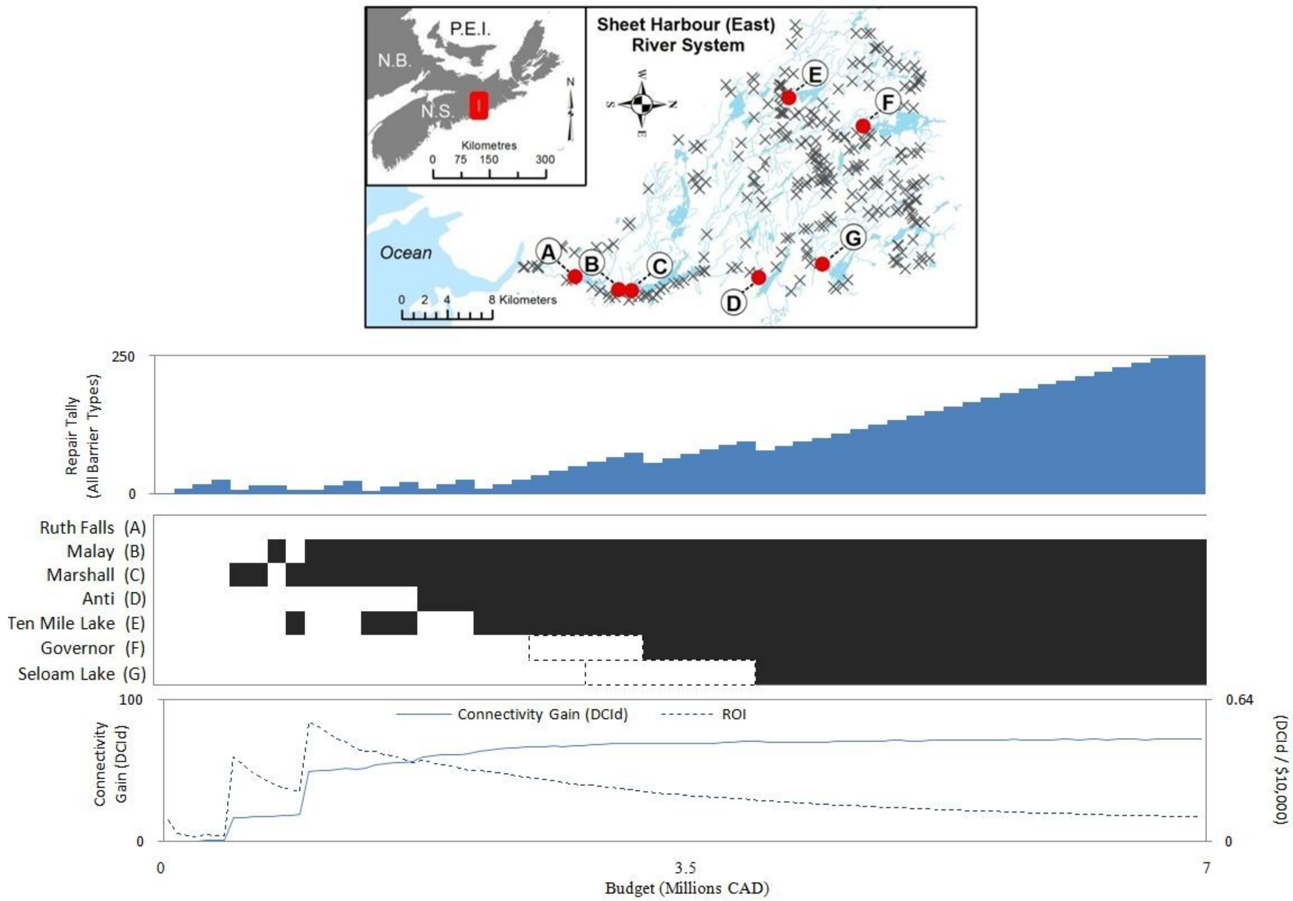

Figure 39: Results of the directed model, 'Area No Stillwater' network quantification method, for the Sheet Harbour system highlighting appearance of dams. corresponding jumps in total number of barriers appearing in optimal decision sets, DCI_d gains per 10,000 CAD versus budget, and connectivity gains versus budget. Dashed cell borders highlight occasions where dams were affordable but did not appear in optimal decision sets. Hatched cell shading indicates a 'half-repair' (i.e., upstream or downstream connectivity restoration project only).

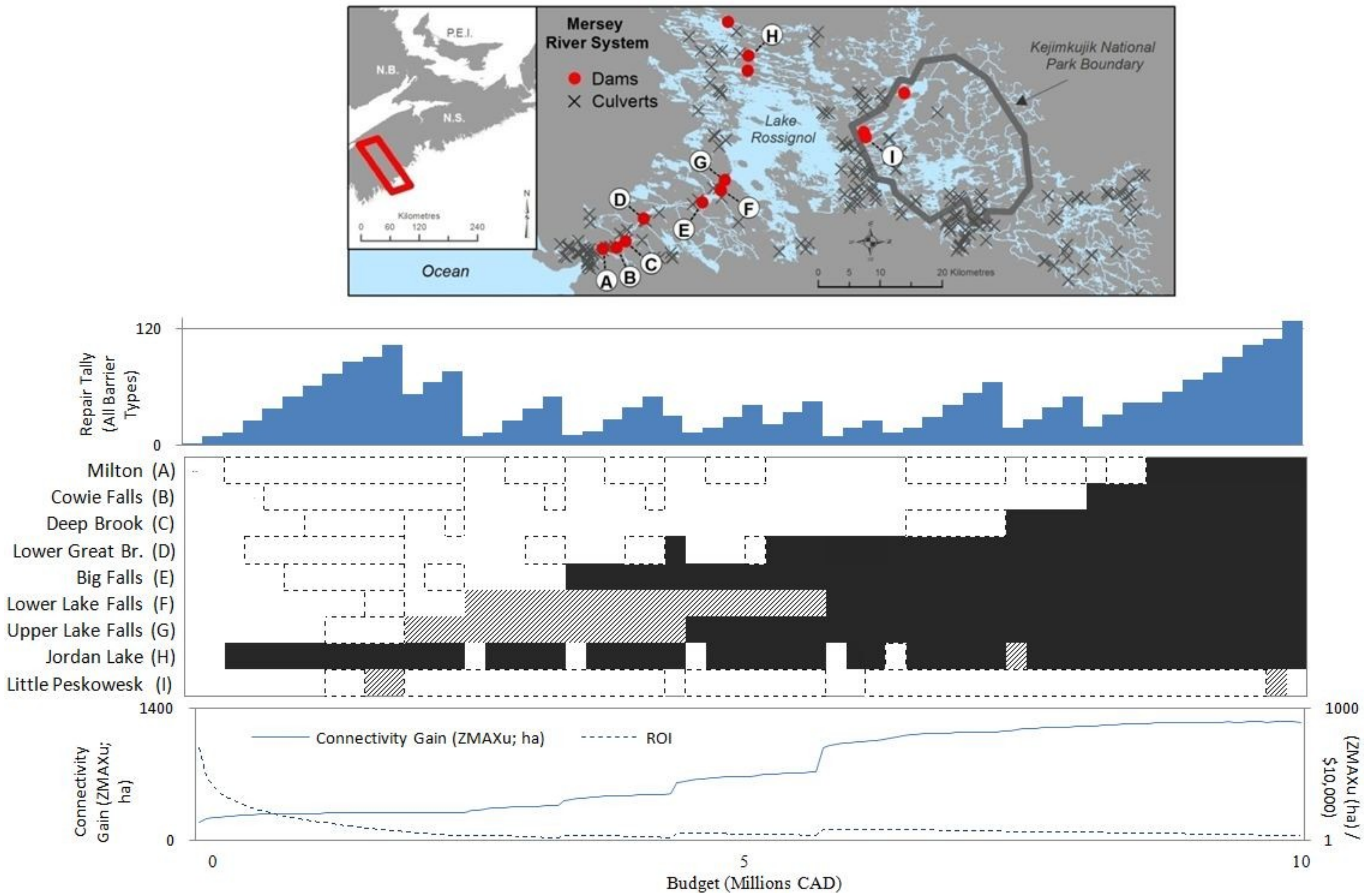

Figure 40: Results of the undirected model, 'Area No Stillwater' network quantification method, for the Mersey system highlighting appearance of dams, corresponding jumps in total number of barriers appearing in optimal decision sets, connectivity gains per 10,000 CAD versus budget, and connectivity gains versus budget. Dashed cell borders highlight occasions where dams were affordable but did not appear in optimal decision sets. Hatched cell shading indicates a 'half-repair' (i.e., upstream or downstream connectivity restoration project only).

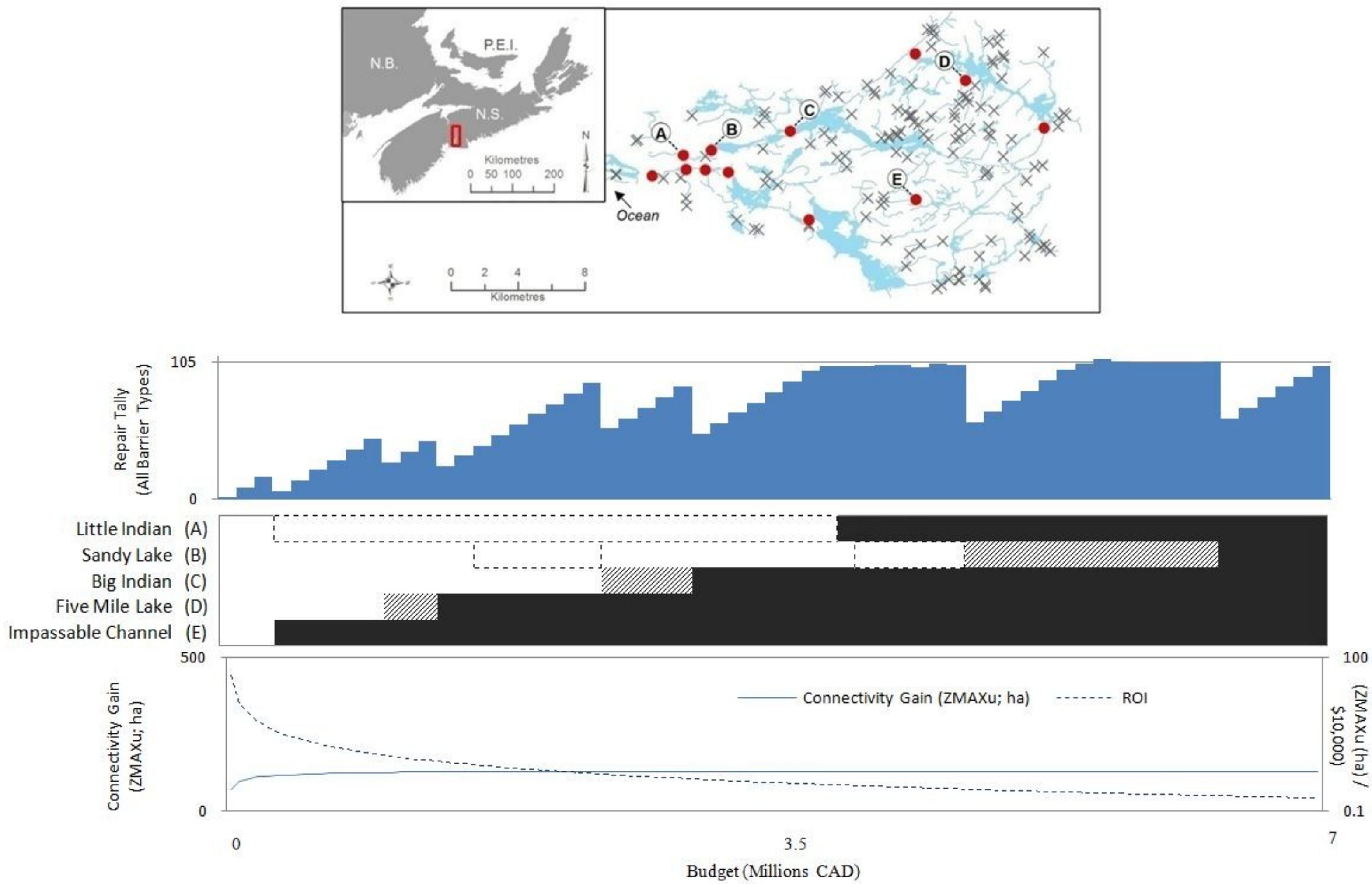

Figure 41: Results of the undirected model, 'Area No Stillwater' network quantification method, for the St. Margaret's Bay system highlighting appearance of dams, corresponding jumps in total number of barriers appearing in optimal decision sets, connectivity gains per 10,000 CAD versus budget, and marginal gains versus budget. Dashed cell borders highlight occasions where dams were affordable but did not appear in optimal decision sets. Hatched cell shading indicates a 'half-repair' (i.e., upstream or downstream connectivity restoration project only).

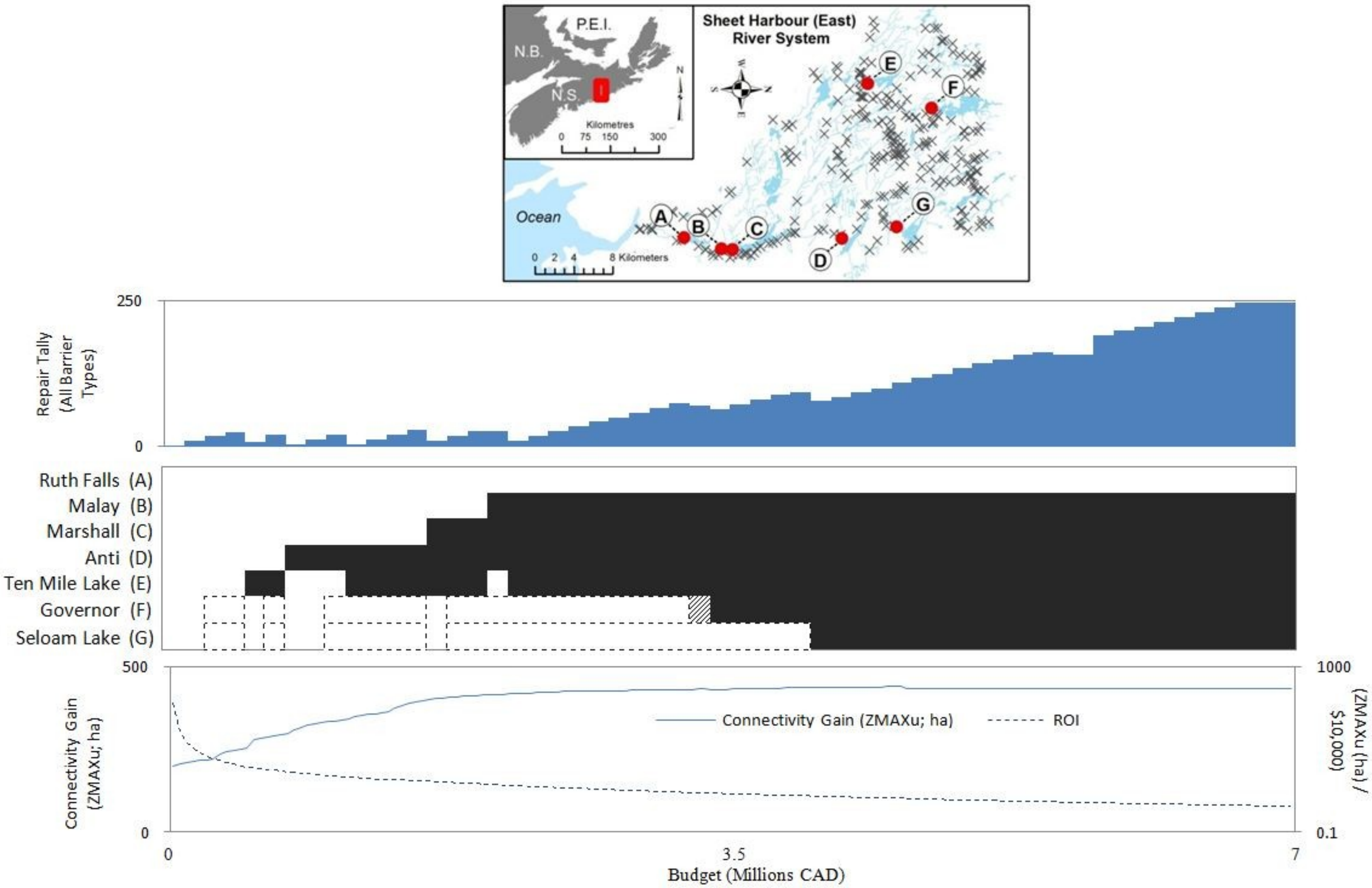

Figure 42: Results of the undirected model, 'Area No Stillwater' network quantification method, for the Sheet Harbour system highlighting appearance of dams, corresponding jumps in total number of barriers appearing in optimal decision sets, connectivity gains per 10,000 CAD versus budget, and marginal gains versus budget. Dashed cell borders highlight occasions where dams were affordable but did not appear in optimal decision sets. Hatched cell shading indicates a 'half-repair' (i.e., upstream or downstream connectivity restoration project only)..

4.5 Cumulative Effects Analysis Results

Separate analyses were conductedusing the 'Area No Stillwater' quantification method with project costs controlled for. The results of the cost-controlled analyses revealed that (1) individual culverts outweigh the benefits of individual dams, even when their costs were assumed equal (though this was infrequent), (2) results were often non-nested (i.e., barriers appearing in an optimal decision set may not appear in the next incremental decision set), and (3), in the case of the Mersey system, connectivity gains per dollar and marginal gains versus budget curves deviated (Figure 46 & Figure 47) from previously reported patterns of declining connectivity gains per dollar and marginal connectivity gains as budgets increased (O'Hanley & Tomberlin, 2005; Zheng et al., 2009; O'Hanley, 2011). In the results of the directed model, culverts appeared in the fifth, sixth, and eighth priority position, outweighing the benefits of alternative dam repairs (Figure 43). In the results of the undirected model, culverts appeared in the results in first, second, third, sixth, seventh, and eighth positions (Figure 43). The results of analysis of the Mersey system using the undirected model revealed that a single culvert was prioritised above any single dam and that this result was non-nested; the combined removal of two dams outweighed the benefit of the removal of the first-place culvert (Figure 44) .The mitigation of a single culvert from St. Margaret's Bay system improved undirected connectivity more than any single dam (except the 'Impassable Channel' barrier representing an unconventional project facilitating an interbasin transfer) with this result being non-nested – at the next incremental budget the combined removal of a dam and a second culvert outweighed the benefit of the first culvert (Figure 45). The returns in terms of connectivity per dollar and absolute connectivity gains versus budget curves generally lacked the pronounced jumps present that analyses including costs exhibited, though the results of the cost-controlled analysis using the directed model for the Mersey system showed positive slope in connectivity returns per dollar for the first few decision sets (Figure 46 & Figure 47).

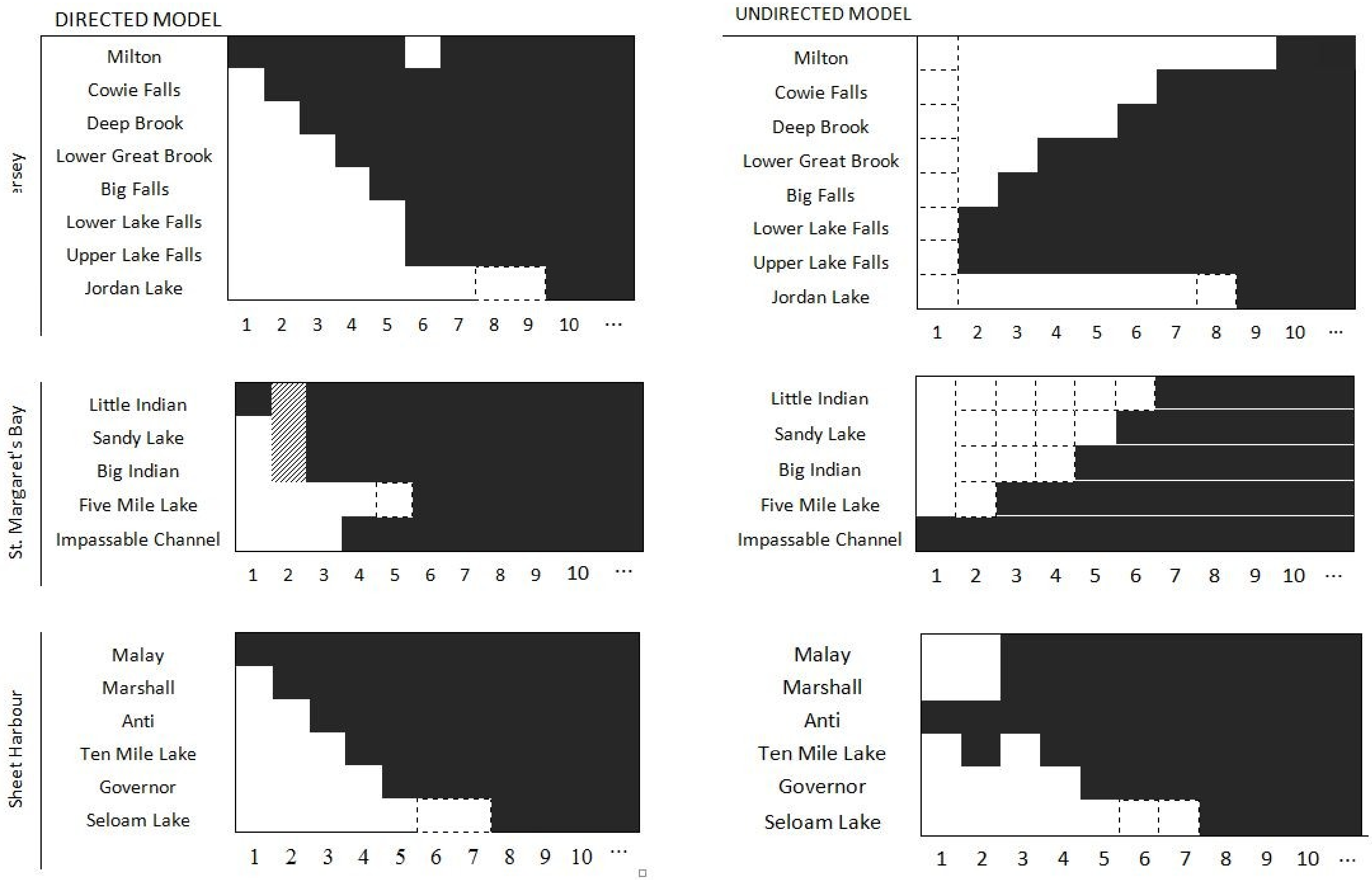

Figure 43: Presence of dams in optimal decision sets for the optimisation treatment 'Area No Stillwater' with all costs set equal. X-axis shows number of barriers repaired, shaded cells indicate dams appeared in optimal decision output, hatched cells indicate a unidirectional restoration project (half-repair), and dashed outlines indicate dam was affordable but did not appear in optimal output.

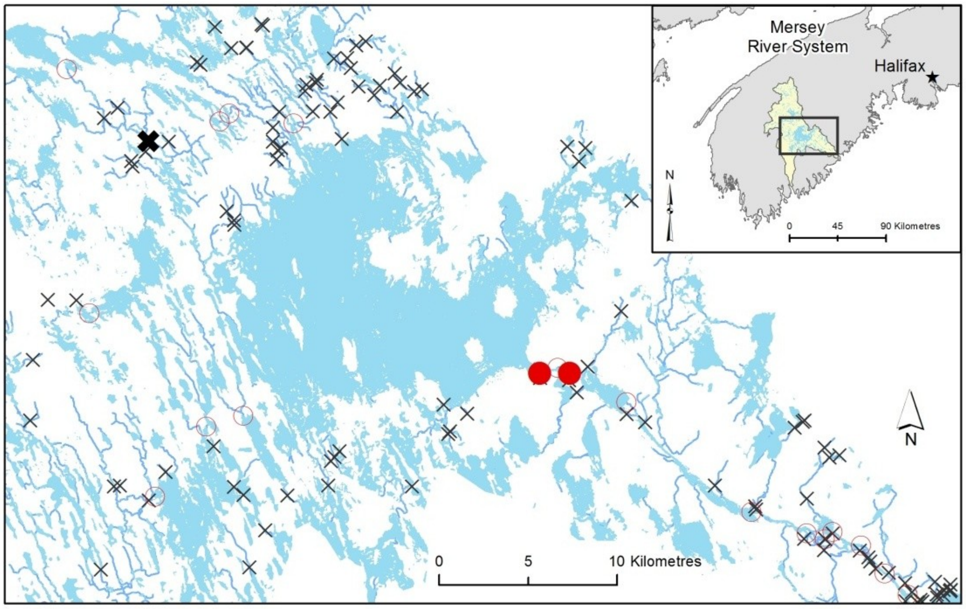

Figure 44: Map of culverts (X's) and dams (circles) with results of the cost-controlled analysis of the Mersey system using the undirected model. The highlighted culvert outweighed all other dams when only one barrier was allowed in optimal decision set. Two solid red dams in combination out-weighed culvert when two barriers were allowed in optimal decision set.

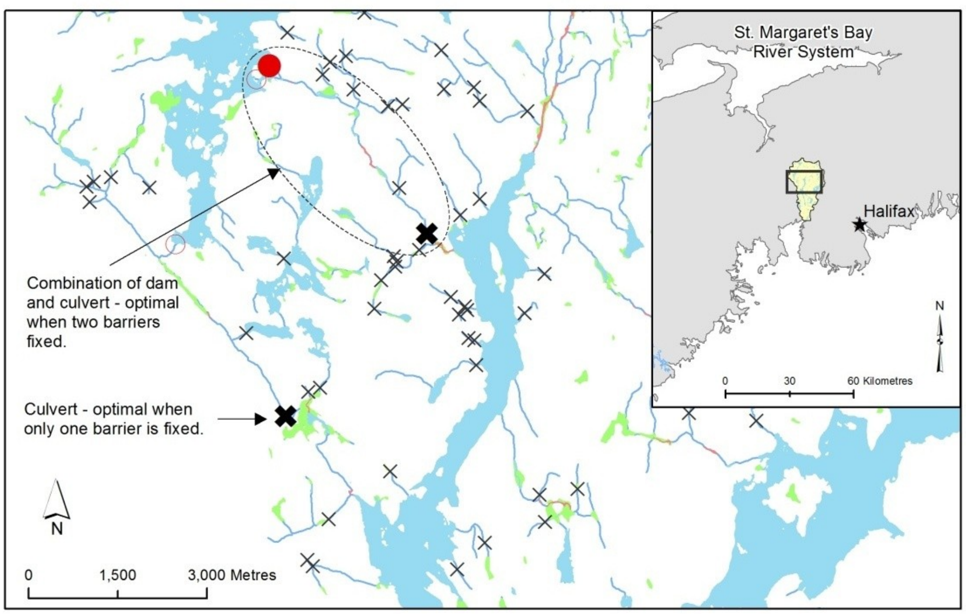

Figure 45: Map of culverts (X's) and dams (circles) with results of the cost-controlled analysis of the St. Margaret's Bay system using the undirected model. A single culvert outweighed all dams when only one barrier was allowed in the optimal decision set. A combination of a second culvert and dam appeared in the optimal decision set when two barriers were allowed.

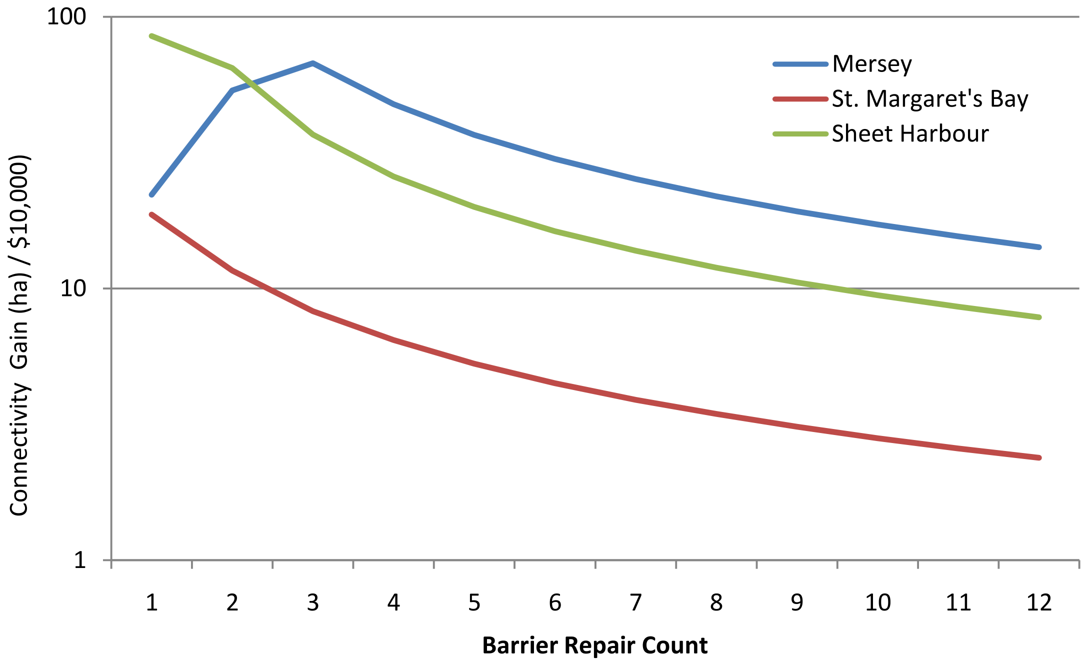

Figure 46: Connectivity gains per 10,000 CAD ($\log_{10}$ scale) versus barrier repair count for the cost-controlled analysis using the directed model.

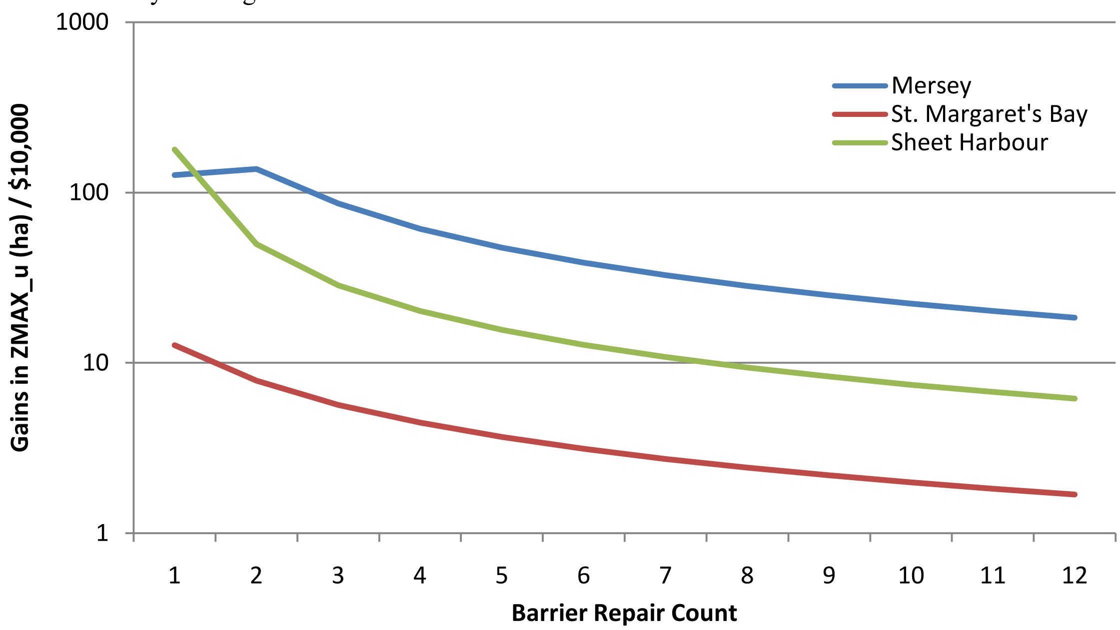

Figure 47: Connectivity gains per 10,000 CAD ($\log_{10}$ scale) versus barrier repair count for the cost-controlled analysis using the undirected model.

Prioritisations were not always nested at incremental budgets, even among barriers with identical costs; individual culverts appearing at lower budgets did not always appear at higher budgets.An example was found in the Sheet Harbour river system. At low budgets for the undirected model, where no dams are prioritised, the culverts at two incremental budgets display non-nestedness (Figure 48).

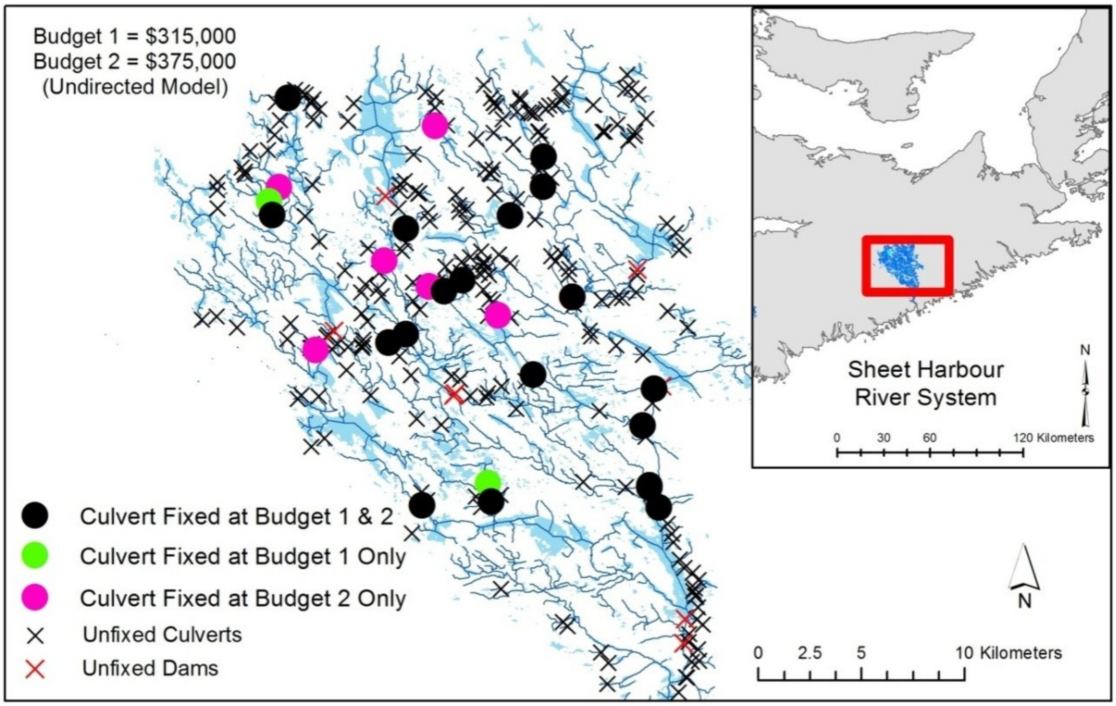

Figure 48: Two incremental budgets where culverts only (identical costs) are chosen by the undirected model. Some culverts chosen at the lower budget were not included in the higher budget, demonstrating non-nestedness of priorities.

4.6 Culvert Permeability Sensitivity Analysis

Results suggest that culvert permeability would have little impact on overall connectivity gains achieved from optimal barrier repair scenarios. That is, under a scenario of imperfect information wherein barrier permeabilities are estimated crudely by a 'best guess', the priorities identified by optimisation were estimated to yield only slightly sub-optimal results as compared to a scenario wherein decisions were made based on 'perfect information.' For the directed model, the largest estimated percentage of budget wasted through decisions based on the ‘best guess’ scenario range was 0.22 - 2.35% (Table 8: Results of Culvert Permeability Sensitivity Analysis using the Directed Model.). The worst-case connectivity gain sacrifice for the directed model was 0.22-2.41% (Table 8). For the undirected model, the maximum percentage of budget wasted and connectivity sacrifice were higher than results of the directed model at 1.46 - 6.08% and 1.48 - 6.47%, respectively (Table 9).

Table 8: Results of Culvert Permeability Sensitivity Analysis using the Directed Model.

MODEL:	Directed		UNITS OF MEASURE:	Metres		
SYSTEM	**Mersey**		**St. Margaret'sBay**		**Sheet Harbour**	
	BUDGET 1	**BUDGET 2**	**BUDGET 1**	**BUDGET 2**	**BUDGET 1**	**BUDGET 2**
Budget	6255	9015	4500	6015	2475	3975
AVG MIP GAP %	0.11	0.10	0.71	0.85	0.69	0.70
Randomisations	30	30	30	30	30	30
Omitted Randomisations	0	0	0	0	2	3
N	30	30	30	30	28	27
AVG MIP GAP % Remaining	0.11	0.10	0.71	0.85	0.61	0.63
AVG Money Wasted ($1,000s)	8.27	13.73	40.05	29.10	25.94	42.47
Variance Money Wasted ($1,000s)	13.01	7.21	206.65	135.39	126.11	265.09
Maximum Money Wasted ($1,000s)	**16.45**	**19.67**	**68.28**	**53.63**	**58.15**	**81.46**
AVG Money Wasted (% of Budget)	0.13	0.15	0.89	0.48	1.05	1.07
Variance Money Wasted (% of Budget)	0.00	0.00	0.10	0.04	0.21	0.17
Maximum Money Wasted (% of Budget)	**0.26**	**0.22**	**1.52**	**0.89**	**2.35**	**2.05**
AVG Absolute Gain Sacrifice (Quan. Units)	73.87	147.25	78.04	82.17	544.72	575.92
Variance Absolute Gain Sacrifice (Quan. Units)	10.40	8.35	7.92	10.88	561.57	493.30
Maximum Absolute Gain Sacrifice (Quan. Units)	**147.37**	**210.73**	**133.10**	**151.77**	**1225.61**	**1107.66**
AVG Gain Sacrifice (% Best-Guess Gain)	0.13	0.15	0.90	0.49	1.06	1.08
Variance Gain Sacrifice (% Best-Guess Gain)	0.00	0.00	0.04	0.04	0.18	0.18
Maximum Gain Sacrifice (% Best-Guess Gain)	**0.26**	**0.22**	**1.54**	**0.90**	**2.41**	**2.09**

Table 9: Results of Culvert Permeability Sensitivity Analysis using Undirected Model.

MODEL:	**Undirected**		**UNIT OF MEASURE:**	**Hectare**		
SYSTEM	**Mersey**		**St. Margaret's Bay**		**Sheet Harbour**	
	BUDGET 1	**BUDGET 2**	**BUDGET 1**	**BUDGET 2**	**BUDGET 1**	**BUDGET 2**
Budget	375	675	435	675	675	2715
AVG MIP GAP %	1.97	27.74	2.70	1.92	21.71	3.37
Randomisations	30	30	30	30	30	30
Omitted Randomisations	0	10	3	0	10	0
N	30	20	27	30	20	30
AVG MIP GAP % Remaining	1.97	23.57	1.80	1.92	18.33	3.37
AVG Money Wasted ($1,000s)	11.38	11.00	7.41	9.23	23.63	29.50
Variance Money Wasted ($1,000s)	30.99	28.17	12.26	9.84	61.60	53.41
Maximum Money Wasted ($1,000s)	**22.78**	**29.69**	**14.60**	**15.22**	**36.15**	**39.53**
AVG Money Wasted (% of Budget)	3.03	1.63	1.70	1.37	3.50	1.09
Variance Money Wasted (% of Budget)	2.20	0.62	0.65	0.22	1.35	0.07
Maximum Money Wasted (% of Budget)	**6.08**	**4.40**	**3.36**	**2.25**	**5.36**	**1.46**
AVG Absolute Gain Sacrifice (Quan. Units)	4.15	2.51	1.36	1.20	2.30	2.43
Variance Absolute Gain Sacrifice (Quan. Units)	3.12	0.27	0.41	0.16	0.55	0.36
Maximum Absolute Gain Sacrifice (Quan. Units)	**7.22**	**3.45**	**2.66**	**1.96**	**3.47**	**3.36**
AVG Gain Sacrifice (% Best-Guess Gain)	3.15	1.66	1.74	1.39	3.64	1.10
Variance Gain Sacrifice (% Best-Guess Gain)	0.68	0.68	0.23	0.23	0.08	0.08
Maximum Gain Sacrifice (% Best-Guess Gain)	**6.47**	**4.60**	**3.47**	**2.31**	**5.66**	**1.48**

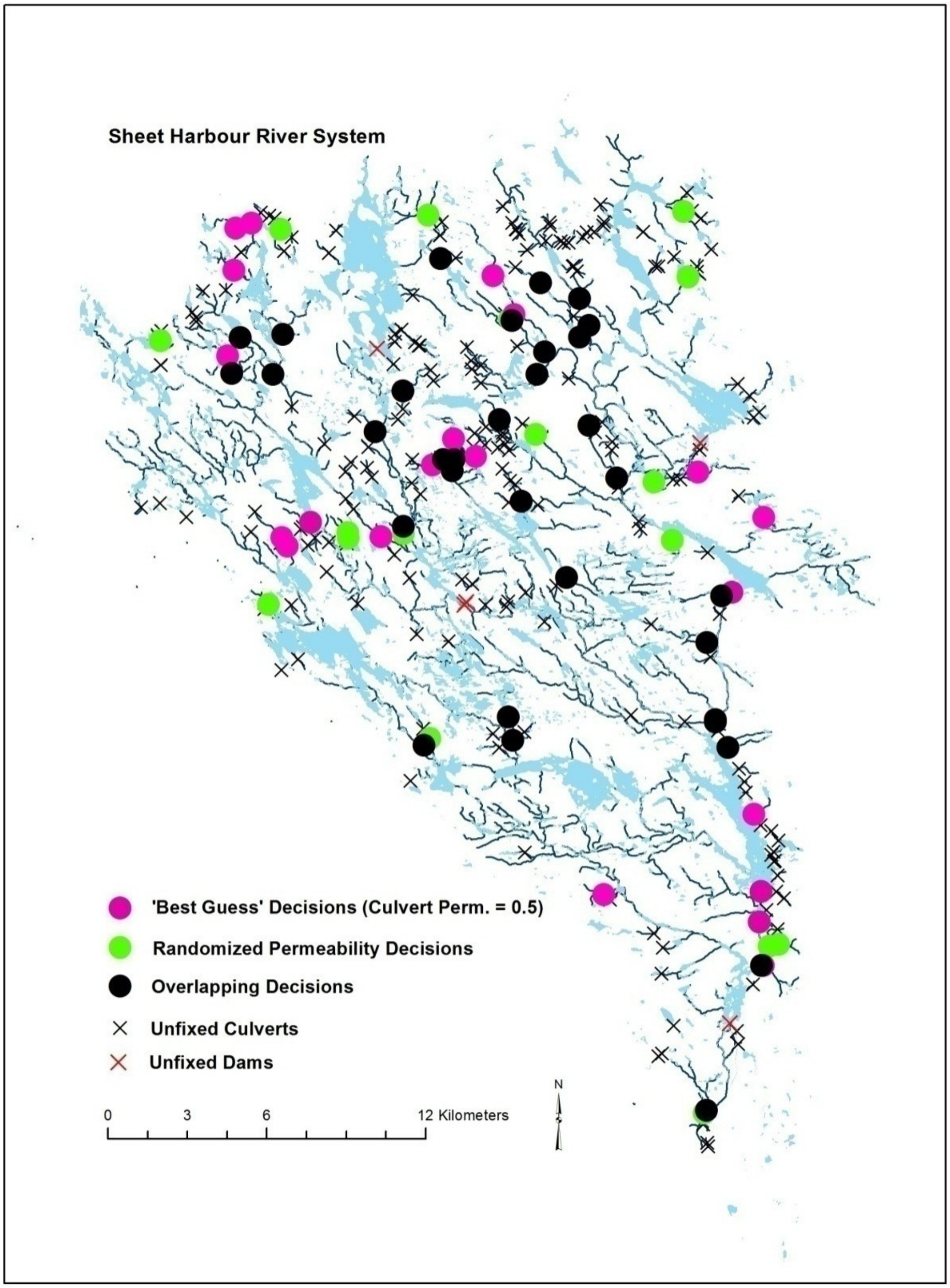

Figure 49: Map of the Sheet Harbour System with overlapping decisions between 'best guess' decisions and 'randomised permeability' decisions for a single randomisation. The difference in terms of connectivity gains and budget expenditure efficiency between the 'best guess' and 'randomised permeability' decisions were compared.

CHAPTER 5: DISCUSSION

5.1 Model Tractability

Maximising undirected connectivity proved to be a significantly less tractable problem than maximising directed connectivity (Table 4 & Table 5). This was an expected result, as solving the directed model is a sub-problem of the undirected model. In response to intractability, previous studies have presented heuristic or dynamic programming methods that reduce the size of the problem (e.g., O'Hanley & Tomberlin, 2005; O'Hanley, 2011). Similar supplementary heuristics could also be developed and employed along with the models presented here to improve tractability of analyses on systems with more barriers than the ones investigated in this study.

5.2 Dendritic Assumption

Both the directed and undirected models are formulated assuming that river networks are DENs, a framework that has gained traction in recent years (Fagan, 2002; Grant & Fagan, 2007; Schick & Lindley, 2007; Cote et al., 2009; Neeson et al., 2011; Peterson et al., 2013). Of course, river systems frequently do not conform to the dendritic pattern, exhibiting natural braiding patterns and inter-basin transfers.Inter-basin transfers frequently arise due to human activity, with the intention of solving water distribution and supply problems (e.g., Grant et al., 2012) or as a result of hydropower development (as in the Mersey and St. Margaret's Bay river systems in this study). River braids and loops are quite easily accommodated by the models and general approach presented here, with only minor edits to the network, though inter-basin transfers must each be considered independently. In the directed model, if there are two possible routes to and from the ocean, then connectivity restoration on one route must be considered in a separate analysis from any other route, rather than one analysis considering both. In contrast, the undirected model can accommodate inter-basin transfers, given that the path to and from the ocean is not of special concern. The one requirement of both models in their current form is that only one path occurs between any given network barrier and any other barrier.

5.3 Stream Width Model

Refinement of models that exploit known relationships between data available for wide geographic extents and stream size address the practical problem of estimating the surface area of streams and facilitate the integrated study of streams and lakes (Rosenfeld & Jones, 2010). The stream width model and associated analysis results support previous observations of a positive relationship between upstream network (or drainage area) and stream size (Betz et al., 2010; Hughes et al, 2010). Although the observed relationship is weak ($R^2 = 0.42$) and deteriorates as network distances from headwaters increase (especially >50 km), the model and methods presented here show both practical and theoretical promise, especially considering that nearly all river segments in need of width estimation had less than 50 km of total upstream network (95.95% by count, 97.17% by length). In addition, the observation that the relationship between 'distance to headwaters' and stream width deteriorates as the former increases is theoretically valuable, as it is consistent with the hypothesis that lake or reservoir effects contribute to variation in stream size (see Rosenfeld & Jones, 2010, for a review).

In the context of this study, a rough estimate of stream width was adequate to compare optimisation budget curves among the different treatments. However, in other studies it should be used with caution. The risk of over-estimating stream widths was mitigated by establishing an upper stream width limit of 27 m. In addition, >75% of stream width estimates were <6 m. Margins of error are unknown, as ground-truthed validation was not carried out and validation was conducted with existing geospatial data.

5.4 Network Quantification Method and Systemic Connectivity

The pronounced differences in the connectivity assessment results of the Mersey system, especially for undirected connectivity between the 'area' ($DCI_p = 64.37$) and 'area no stillwater' ($DCI_p = 27.28$) treatments, show that habitat quality weightings can certainly alter systemic connectivity assessments. This is an important consideration when taking a multi-species approach to prioritisation. For example, the catadromous American eel is not known for a great preference in habitat type, found in comparable abundance and size in streams, lakes, reservoirs, and rivers of the three rivers examined (D. Thompson, personal communication, May 22, 2012) with dispersal models being a better predictor of presence and abundance than habitat suitability (Smogor et al., 1995). In contrast, the Atlantic salmon has very particular habitat preferences and, for spawning purposes, will generally not utilise lentic waterbodies. Notable differences between 'area' and 'length' treatments were also observed. The Sheet Harbour undirected connectivity assessment from the 'length no stillwater' ($DCI_p = 37.62$) and the 'area no stillwater' ($DCI_p = 51.14$) treatments, for example, illustrates that quantification method can affect longitudinal connectivity assessments.

Margins of error are difficult to calculate and future studies may look to empirically quantify the statistical significance of the differences between connectivity assessments found here. Length and area measures of river features were derived from aerial photos, digital elevation models, and hydrographic drainage models (GeoNova, 2012), though there were no metadata available with an accuracy standard. Because of difficulty ascertaining accuracy and error margins of the length and area measures used, no robust analysis to determine if differences between connectivity measures were statistically significant was conducted. Further development of the directed and undirected models could incorporate uncertainty and margins of error into the calculations, perhaps through stochastic optimisation methods. Despite this shortcoming, the larger differences found were deemed adequate to provide some sense of the variability as a starting point for future research.

5.5 Optimisation Results

Network feature quantification method did not have a drastic effect on the slope or shape of connectivity gains per dollar versus budget curves, though some differences were apparent. For example, the results of optimisation in the Mersey system for the undirected model showed markedly different connectivity gains per dollar versus budget and connectivity gains versus budget curves for the 'area no stillwater' treatment versus the other methods of network quantification (Figure 31-Figure 34). Inter-system rankings, in terms of connectivity returns per dollar spent, also changed slightly for this system between treatments. Similar differences in inter-system rankings were observed between the Mersey system and others in the results of the ‘areas’ quantification treatment for the directed model (Figure 27-Figure 30). The Mersey system results appeared to be particularly sensitive to area quantification method, likely due to the large reservoirs (e.g., Lake Rossignol) present.

Inter-system rankings were affected by the method of assessing longitudinal connectivity. The relatively small St. Margaret’s Bay network, for example, often ranked much higher relative to the other two systems when the DCI was used as the metric of connectivity rather than absolute permeability weighted network. The choice of relative or absolute methods of longitudinal connectivity assessment may therefore be an important consideration in inter-system budget allocations.

The observation of broad scale jumps in marginal connectivity gains in the results of the directed model (Figure 27-Figure 30) can be attributed to the relatively large differences in repair costs and longitudinal connectivity impact between the barriers examined in this study; as budget increments and axis scales are selected to make gains to connectivity from culverts visible, those gains from dams appear exaggerated. In a study where variability in costs and benefits was less, such jumps would likely not be as pronounced. The presence of these jumps in connectivity gains per dollar versus total budget emphasises the need to conduct optimisation over a suite of budget amounts, to identify thresholds wherein relatively trivial increases in budget may yield significant gains in expenditure efficiency.

The results of the undirected model generally conformed to a pattern of decreasing connectivity gains per dollar as budgets increased while results of the directed model were not consistent, often showing overall trends of positively sloped gains per dollar (i.e., *economies of scale*). Overarching trends of non-negative slopes are an expected result if sets of barrier mitigation projects are interacting to yield connectivity gains greater than the sum of the gains of the projects taken individually (an example of non-additive and synergistic effects); as budgets increase, more projects can generally be afforded, thus increasing the chances of synergistic interactions between projects.The St. Margaret's Bay system, for example, with three key barriers - Little Indian Lake dam, Sandy lake dam, and Big Indian Lake dam – has no access from the ocean to the western side of the river system (Figure 9). The first two barriers encountered while traveling upstream from the ocean (and, to a lesser extent, the third barrier), act as one large barrier. The large collective impact to directed connectivity occurs because (1) the barriers are located near the outflow of the system and (2) they are spatially distributed in a clustered (versus uniform or random) pattern. The expected returns to systemic connectivity of the mitigation of any of these barriers taken individually are negligible, though the cumulative effect of the mitigation of all of them is significant. This is an important observation for two reasons. First, SR and stepwise SR methods of prioritisations are ill-equipped to quantify combinatorial effects of barrier mitigation. Second, several previous studies have observed the impact to longitudinal connectivity of the addition or removal of barriers using hypothetical networks and have assumed a random or uniform distribution of barriers (Cote et al., 2009; Padgham & Webb, 2010). Padgham and Webb (2010), for example, found that multiple structural modifications to longitudinal connectivity produced near-additive responses, though they noted that if such modifications (i.e., barriers) were located close to each other on the network then the responses may be non-additive. In addition, Padgham and Webb (2010) asserted that the occurrence of non-additive effects would either be acceptably negligible or incorporable on a case-by-case basis, treating clustered groups of barriers as one. However, such clustered groupings are likely common due to spatial concentration of anthropogenic development (Jeong et al., 2010) and, in most cases, manual review of a network to assess costs and benefits of removal of each set of groupings may often be impractical, given the number of barriers and the difficulty in determining which groupings constitute clusters and which do not.

5.6 Meta-analysis: Culverts versus Dams

It seems two attributes of dams observed in this study contributed to their dominance in directed longitudinal connectivity restoration priorities: (1) the tendency of positioning of dams near the system sinks (i.e. outflows), and (2) the tendency of dams to impound a greater amount of river network than culverts, even when stillwater is omitted from consideration (Figure 35). Delays in dam appearance beyond the budget where they became affordable (after the cost of other higher-ranked barriers was factored in) were infrequent in the results of the directed model, and their appearance generally corresponded with spikes in connectivity gains per dollar and absolute connectivity gains (Figures 37-39). The correspondence of such spikes with dam appearance in optimal decision sets along with the disappearance of such spikes in the cost-controlled analysis suggests the high mitigation cost of some barriers is responsible for these spikes, rather than synergism among projects. Conversely, positioning of dams is likely to help explain the frequency in which culverts were selected over dams in the optimal prioritisations identified by the undirected model; positioning near the outflow distances them from the centre of the network which is likely to be important for undirected connectivity. It is worth noting that measures of network centrality used in graph theory may therefore be helpful in identifying river segments of importance for undirected connectivity (see Estrada and Bodin, 2008). It also seems that a particular characteristic of undirected connectivity contributes to this occurrence: there are $n + 1$ (where n is the number of barriers) times more undirected subnetworks to choose from when seeking to maximise undirected connectivity whereas directed connectivity must maximise connectivity on the single network connected to the sink. Therefore, there are far more opportunities over the extent of the network to maximise the single largest undirected subnetwork while avoiding the expense of dam mitigation.

Setting aside cost, culverts were found to impair longitudinal connectivity far less than dams, both individually (with some outliers – e.g., Figure 45) and as a group. Aggregate estimates of impounded upstream network showed that culverts impounded less river network than dams, with some variation found between the quantification methods used (Figure 35). Simulated removal of both barrier types as groups supported the indications given by impounded network results; i.e., significantly greater gains to longitudinal connectivity would be achieved via dam mitigation as compared to culvert mitigation

(Table 7). It is important to note that these results were found even when stillwater was omitted from consideration, removing any potential 'reservoir effect' from skewing prioritisations in favour of dams. Caveats are required, however; the culvert inventory used is potentially a significant underestimation of the actual number present and the 50% barrier permeability estimation may also be an underestimate. In addition, a single culvert was prioritised above all other dams in the results of the undirected model both the Sheet Harbour and Mersey systems (Figure 44 & Figure 45).

5.7 Culvert Permeability Sensitivity Analysis

It is not surprising that the sensitivity to culvert permeability was higher when seeking to restore undirected compared to directed longitudinal connectivity, as culverts were found to play a more significant role in reducing undirected longitudinal connectivity. Despite being greater, the sacrifices due to imperfect information on culvert permeability did not exceed 4% even for the undirected model, both in terms of connectivity and budget allocation efficiency. This result is likely to vary for systems containing fewer hydropower dams and may depend on the network quantification method used, though 'area no stillwater' was selected for this analysis because this method was hypothesised to minimise chances of underestimating the effects of culverts.

For future studies, there are several opportunities to improve upon the sensitivity analysis methodology presented here. The approach used herein simulated random permeabilities for all culverts (0.0 - 1.0), found optimal decision sets based on these estimates, compared the difference in systemic connectivity gains between the optimal decisions sets found under randomized permeability to the gains under a 'best guess' permeability of 0.5, and repeated this for 20 – 30 randomisations. First, an improvement could be made by increasing the number of randomisations used (due to time limitations this was not done). Second, different randomisation methods could be applied.For example, rather than randomisation permeabilities from 0.0 to 1.0, they could be randomized as a binary variable (0 or 1). Third, the sensitivity analysis assumes two scenarios and compares tradeoffs between them: (1) conduct optimisation analysis and take action given imperfect information, and (2) investigate all culverts, assess permeability, conduct optimisation analysis, and take action. Alternatively, a method to assess sensitivity to permeability could be conceived where only (1) is assumed, and the difference between the expected connectivity gains are compared to the actual connectivity gains. The method used was chosen because it may help better assess opportunity cost of foregoing culvert surveying and robust permeability modeling and it was outside the scope of this research to conduct actual barrier repairs with robust connectivity assessment before and after. Fourth, and perhaps most importantly, stochastic optimisation methods could be used in which permeability could be incorporated as 'fuzzy' variable, thus robustly accounting for the uncertainty surrounding this variable.

5.8 Cumulative Effects Analysis

The simulated removal of the first three barriers on the Mersey system (Milton, Cowie Falls, and Deep Brook dams) yielded increasing returns in terms of connectivity per unit effort, explained by spatial interdependence of these barriers (Figure 46 &Figure 47). Prioritisations obtained from the undirected model also showed some evidence of synergism between the mitigation of Upper Lake Falls and Lower Lake Falls dams of the Mersey system which, taken altogether, outranked the first-placed culvert (Figure 44). Both circumstances are characterised by barriers impounding relatively large amounts of network and appearing in close proximity to each other.As Padgham and Webb (2010) observed, clustered barrier groupings are likely to result in synergistic interactions between restoration projects, leading to non-additive cumulative effects of restoration that are difficult to predict without the use of combinatorial methods such as optimisation.

Whether clustered spatial arrangement of barriers on river networks is a common occurrence is under-explored. Past studies have suggested barrier clustering to be negligible, easily detectable, and unlikely to adversely affect the effectiveness of restoration priorities arrived at through SR, stepwise SR, and other non-combinatorial methods (Padgham & Webb, 2010). However, results reported here and elsewhere (Jeong et al., 2010) suggest that anthropogenic effects to riverscapes are often spatially concentrated, and clustered arrangements of barriers may thus be more common than previously assumed.

Return-on-investment and connectivity gains versus budget curves lacked pronounced jumps when costs were controlled for, suggesting that high dam-repair costs relative to culverts were responsible for such observations in the cost-included analyses. This can be explained by the high dam-mitigation cost delaying the appearance of dams in output decision sets until they were affordable, causing sudden jumps in connectivity gains versus effort. When costs were omitted from considerations, connectivity gains per unit effort became negative as effort increased and was generally consistent with previous studies (O'Hanley & Tomberlin, 2005; O'Hanley, 2011; Diefenderfer et al., 2012). The frequency of non-nestedness of dam repair was also reduced, with only two occasions observed (**Error! Reference source not found.**). In the results of the undirected model for the Sheet Harbour system, neither Malay nor Mashall dams individually out-prioritised the Ten Mile Lake dam, though when considered together they were able to, causing the Ten Mile Lake dam to disappear from priorities.The prioritisations of the Sheet Harbour system from the directed model showed that Milton was a priority when six barriers were prioritised but not when seven were - Milton was outranked by the combined and synergistic effect of Upper and Lower Lake Falls dams. It should be noted that selection of culverts for mitigation may also be playing a role, though analysis of nestedness of culverts in output prioritisations was not undertaken. That these two observations of non-nestedness correspond to a normal, declining trend of connectivity returns per unit effort versus budget (Figure 46 & Figure 47) demonstrates that non-nestedness does not always correspond to unusual or irregular trends in connectivity returns per unit effort versus budget.

CHAPTER 6: CONCLUSION

In response to severe, anthropogenic fragmentation of rivers worldwide, many billions of dollars have been spent in the past two decades to mitigate the detrimental effects of disrupting riverine connectivity. Despite these efforts, the size of the problem still far exceeds the resources available and so there is a need to efficiently prioritiserestoration efforts. However, common techniques of cost-benefit analysis and prioritisation are often inadequate to deal with the large number of restoration options available and the spatial interdependence of projects. Furthermore, assessing the benefits of restoration projects at the riverscape scale is also a major challenge.

To help address the issues faced in systemic river restoration planning, two optimisation models weredeveloped and demonstrated as part of this thesis research. They are designed to maximise longitudinal connectivity of river networks given a limited budget. The incorporation of the directional component distinguishes the two models, reflecting the contrasting ecological needs of resident and marine-migratory species. Incremental but notable improvementswere made over existing models, lowering the level of expertise needed and accounting for the variable degree to which barriers can impede connectivity. They were embedded within GIS software which allowed for easy visualisation of prioritisations and made results more transparent to the user. Application of these models on three river systems in Nova Scotia, Canada, demonstrated that these models coupled with GIS are a powerful tool for river restoration planning.

The systems selected for analysis contained both hydropower dams and numerous culverts found at road crossings. These types of systems were chosen to help answer the question of whether combinations of culverts could outrank dams in prioritisations. Results of barrier removal simulation and the meta-analyses of optimisation results suggest that combinations of culverts can sometimes outrank dams in prioritisations and that this is more common when the aim is to maximise undirected connectivity. Results also show that on river systems such as these, with large hydropower dams, marginal gains often increase as budgets increase, contrary to findings of other studies (though some marginal increases could be explained by thresholds exaggerated by the scale of the graph). Increasing marginal gains were observed especially in the results of the directed model. This is an important observation, as it suggests that: (1) selection of budget can affect efficiency of restoration efforts and (2) the choice of objective, whether it be to restore directed or undirected connectivity, is important.

The assessment of barrier permeabilities can be expensive and time-consuming. Results suggest that accurate assessment of culvert permeability may be more important when the aim is to restore undirected connectivity than directed connectivity. However, in the worst case scenario the basic sensitivity analyses suggested that uncertainty about culvert permeability did not greatly affect expected optimal gains (<4% sacrifice to connectivity; Table 8 & Table 9).

Due to data scarcity, estimating the size of river network features can be a challenge, especially over wide geographic areas. As a result, most studies to date are limited to employing units of length to represent river network features (e.g., Cote et al., 2009; Bourne et al., 2011; Mount et al., 2011; O'Hanley, 2011; Pini Prato et al., 2011; Anderson et al., 2012; Nunn & Cowx, 2012), though some have used area (Kuby et al., 2005; Zheng et al., 2009) - a more desirable approach from an ecological standpoint (see Hughes et al., 2010). In addition, habitat quality assessments are often undertaken, sometimes for each species of interest (e.g. Kocovsky et al., 2008). The results of this thesis research suggest that methods of quantifying river network and overall systemic connectivity are likely to affect subsequent restoration priorities. Thus far, to my knowledge, this is the first study to report notable variation in systemic connectivity assessment between different quantification methods.

A natural next step in the development of optimisation models for river restoration planning is to propose and test methods of multi-objective optimisation and stochastic optimisation. Such models could be used to investigate the degree to which optimal directed and undirected connectivity restoration priorities are competing or complementary. The question of how similar or dissimilar optimal restoration decision sets are to each other seems at first to be relatively easy to answer through calculating 'decision overlap' – that is, a statistic based the number of barriers in common between prioritisations. However, such an approach does not address the significance of the overlap(e.g., two optimised prioritisations may have a low degree of overlap yet contain a key barrier responsible for the majority of connectivity gains). A better approach of prioritisation comparison is therefore needed. Stochastic optimisation could be employed to better account for uncertainty surrounding various model parameters such as barrier permeability. Furthermore, the temporal dimension could also be incorporated in optimisations, as fluctuations in flow and the quantity and quality of water strongly affects longitudinal connectivity (Pringle, 2001).

The approach taken in this thesis, for demonstration purposes, was simply to exclude lentic waterbodies to de-emphasise these areas as habitat for certain species. To effectively prioritise river restoration efforts for multiple species, efforts to qualitatively weight river network features based on habitat suitability (e.g., Kocovsky et al., 2008; Kocovsky et al., 2009; Zheng et al., 2009; Wang et al., 2013), dispersal models (e.g., Smogor et al., 1995; Schick & Lindley, 2007; Pépino et al., 2012), or other such methods should be used. Future studies could further refine methods used for weighting river network by habitat quality, which could then easily be incorporated into the optimisation models. In addition, permeability weightings could be adjusted to better reflect the connectivity needs of different species, populations, or life stages of various aquatic species. The directed model is useful in optimising upstream and downstream connectivity and thus able to accommodate both catadromous and anadromous life strategies. The method utilised here weighted upstream and downstream connectivity equally by assigning each a weighting of 50% of total upstream-downstream connectivity. To target a particular species or life strategy, either upstream or downstream directed connectivity could be emphasized as appropriate by changing the permeability weightings.

A foreseeable extension of the modeling approach employed here could be to incorporate all systems into one analysis. This could be done simply by extending the network from the sink of each system to a common node. An approach such as this would be useful in scenarios where funds are allocated for longitudinal connectivity restoration across multiple river networks. Conversely, in scenarios in which particular subnetworks have been identified for restoration action or conservation over others (e.g. Moilanen et al., 2008; Hermoso et al., 2011; Nel et al., 2011), optimisations of restoration options could be conducted for these subnetworks in isolation, by temporarily disconnecting all other subnetworks.

Characterising cumulative effects of anthropogenic development and restoration actions to longitudinal connectivity as additive or non-additive is a priority (e.g., Allan, 2004; Jansson et al., 2007; Diefenderfer et al., 2011; Diefenderfer et al., 2012; Segurado et al., 2013). Non-additive effects can greatly reduce the effectiveness of simple prioritisation methods, such as SR. The presence of synergies in aquatic connectivity restoration has been noted in dike-breach scenarios (i.e. lateral connectivity restoration; Diefenderfer et al., 2012), though the cost-controlled analyses performed in this research (Figure 46 & Figure 47) is notable as one of the few in-depth studies to date revealing non-additive cumulative effects of restoration of the longitudinal connectivity of rivers. Compared to alternative approaches, such as repeated randomized sampling of barriers for removal (Diefenderfer et al., 2012), optimisation offers a unique opportunity to assess effects of a stressor given the ability of the approach to virtually guarantee that no combination of restoration projects would yield better results, given available information.

The type of distribution barriers exhibit (i.e., random, uniform, or clustered; see Estrada & Bodin, 2008) will likely determine the type of cumulative effect they have (see Padgham & Webb, 2010), yet few studies to date have been devoted to this. Future application of graph theoretical approaches to the characterisation of river networks appears promising, such as degree- and betweenness-centrality measures (e.g., Estrada & Bodin, 2008; Dale & Fortin, 2010; Peterson et al., 2013). Adaptation of spatial autocorrelation measures to river networks could also be useful, such as Moran's *I* (Moran, 1950) or Geary's C (Geary, 1954) statistics. Future studies could also develop empirical methods to directly quantify non-additive cumulative effects (e.g., by simulating the mitigation of each barrier individually in a set of priorities, calculating the improvement to longitudinal connectivity, and comparing the sum of these improvements to cumulative improvements attained by through mitigation of the entire set). The result of such an approach would also be a test of the relative efficiency of SR versus optimisation, similar to one done in the past (O'Hanley & Tomberlin, 2005).

In tandem with efforts to naturalise river networks and restore aquatic connectivity, our knowledge of river systems has grown substantially. The recent application of network theory in ecology has been particularly profound; rivers are increasingly conceptualised as ecological networks, characterised by dendritic structure and strong directionality. Recent developments from the field of landscape ecology, of such metrics as the DCI (Cote et al., 2009), allow for connectivity assessments at the scale of the riverscape. In addition, computational power has grown exponentially, making optimisation accessible to the everyday user. This thesis research is evidence that the combination of network analysis, optimisation, and GIS is a powerful tool for river restoration planning and shows great promise towards furthering our understanding of river systems.

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APPENDIX A: OPTIMISATION MODELS

A.1 Directed Model (GLPK)

```
param nNodes;
param FirstNod;
param mOptions;
set I; /* barriers set - G */
set O, default {1 .. mOptions};
set Upstream, within I cross I; /* matrix of barriers for
connectivity - G*/
set Options, within I cross O;/* matrix of barriers vs. options -
G */
param dummy{(i,j) in Upstream}, default 1;
table tab_upstream IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKConnectivity.csv":
  Upstream <- [BEID,UpEID], dummy ~ DUMMY;
param perm{ (i,k) in Options} , default 1;
param cost{ (i,k) in Options} , default 100;
table tab_options IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKOptions.csv":
  Options <- [BARRIER,OPTION1], perm ~ PERM, cost ~ COST;
param Zmax{(i,k) in Options} , default 50000000;

param habitat{ i in I} , default 0;
table tabitat_heheh IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKHabitat3.csv":
  I <- [BARRIER], habitat ~ HABITAT;

param Budget, default 1000;

var y{ i in I}, >=0;                  /* optimized acessible
habit above i */
var z{ (i,k) in Options}, >=0;        /*  accessible habit above
i  if option k is chosen*/
var x{ (i,k) in Options}, binary;    /* option choice variables
at node i */

maximize obj:  y[FirstNod];

s.t. HabAbove{i in I}:  y[i] = sum{ k in O: (i,k) in Options}
z[i,k];

s.t.  HabZ{ i in I,  k in O: (i,k) in Options}:  z[i,k] <=  sum{j
in I: (i,j) in Upstream}( perm[i,k] * y[j]) +
perm[i,k]*habitat[i];

s.t.  UpZ{ i in I,  k in O: (i,k) in Options}:  z[i,k] <=
Zmax[i,k]*x[i,k];
```

```
s.t.  SumX{ i in I}:  sum{ k in O: (i,k) in Options} x[i,k] = 1;

s.t.  BudgetCon:  sum { i in I,  k in O: (i,k) in Options}
cost[i,k]* x[i,k] <= Budget;

solve;

printf "          Barrier        Option     \n";
printf {i in I,  k in O: ((i,k) in Options) and (x[i,k] !=0) }:
"%13s   %11s     %12g  \n", i,  k,  x[i,k];

table res1{i in I,  k in O: ((i,k) in Options) and (x[i,k] !=0) }
OUT "CSV" "C:\GunnsModel_REPLACE\Res1.csv": i~Barrier,k~Option,
x[i,k]~OptionChioce;

printf "               \n";
printf "           Budget        Habitat     \n";
printf "   %12g    %12g  \n", Budget, y[FirstNod];

printf "Habitat     \n" > "C:\GunnsModel_REPLACE\ZMaxOutput.txt";
printf y[FirstNod] >> "C:\GunnsModel_REPLACE\ZMaxOutput.txt";

printf {i in I: (y[i] !=0) }: "   Y[i]     %13s     %12g  \n", i,
y[i];
table res3{i in I: (y[i] !=0) } OUT "CSV"
"C:\GunnsModel_REPLACE\Res3.csv": i~Barrier, y[i]~Habitat;

printf {i in I,  k in O: ((i,k) in Options) and (z[i,k] !=0) }: "
z[i,k]    %13s   %11s     %12g  \n", i,  k,  z[i,k];
table res2{i in I,  k in O: ((i,k) in Options) and (z[i,k] !=0) }
OUT "CSV" "C:\GunnsModel_REPLACE\Res2.csv": i~Barrier,k~Option,
z[i,k]~Habitat;

end;
```

A.2 Undirected Model (GLPK)

```
param nNodes;
param FirstNod;
param mOptions;
set I; /* barriers set - G */
set O, default {1 .. mOptions};
set Upstream, within I cross I; /* matrix of barriers for
connectivity - G*/
set Downstream, within I cross I; /* matrix of downstream
barriers - G          NEW */
set Options, within I cross O;  /* matrix of barriers vs. options
- G */
param dummy{(i,j) in Upstream}, default 1;
param dummy_d{(i,m) in Downstream}, default 1;
/* NEW reversed i,m? */

table tab_upstream IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKConnectivity.csv":
  Upstream <- [BEID,UpEID], dummy ~ DUMMY;

table tab_downstream IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKConnectivity.csv":   /* NEW */
  Downstream <- [UpEID,BEID], dummy_d ~ DUMMY;

param perm{ (i,k) in Options} , default 1;
param cost{ (i,k) in Options} , default 100;
table tab_options IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKOptions.csv":
  Options <- [BARRIER,OPTION1], perm ~ PERM, cost ~ COST;
param Zmax{(i,k) in Options} , default 50000000;
param Qmax{(i,k) in Options} , default 50000000;
/* NEW */

param habitat{ i in I} , default 0;
table tabitat_heheh IN "CSV"
"C:\GunnsModel_REPLACE\FIPEX_GLPKHabitat3.csv":
  I <- [BARRIER], habitat ~ HABITAT;


param Budget, default 1000;
param MArea, default 1.E+08;

var y{ i in I}, >=0;                   /* optimized acessible
habit above i */
var z{ (i,k) in Options}, >=0;         /*  acessible habit above
i  if option k is chosen*/
var x{ (i,k) in Options}, binary;      /* option choice variables
at node i */
var w{ i in I}, >=0;
/* NEW */
```

```
var q{ (i,k) in Options}, >=0;
/* NEW */


var iamx{i in I}, binary;
var AMaxMax, >=0;

maximize obj:  AMaxMax;

s.t. HabAbove{i in I}:  y[i] = sum{ k in O: (i,k) in Options}
z[i,k];
s.t.  HabZ{ i in I,  k in O: (i,k) in Options}:  z[i,k] <=  sum{j
in I: (i,j) in Upstream}( perm[i,k] * y[j]) + perm[i,k] *
habitat[i]; /* end modified */
s.t.  UpZ{ i in I,  k in O: (i,k) in Options}:  z[i,k] <=
Zmax[i,k]*x[i,k];
s.t.  SumX{ i in I}:  sum{ k in O: (i,k) in Options} x[i,k] = 1;
s.t.  BudgetCon:  sum { i in I,  k in O: (i,k) in
Options}(cost[i,k]* x[i,k]) <= Budget;

s.t. DownQ{ i in I,  k in O: (i,k) in Options}:  q[i,k] <=
Qmax[i,k]*x[i,k];              /* NEW */
s.t. MaxAMax{i in I, k in O: (i,k) in Options}: AMaxMax >= y[i] +
w[i]- perm[i,k] * habitat[i] + habitat[i] - MArea*iamx[i];
/* NEW */
s.t. BoundAmax{i in I, k in O: (i,k) in Options}: AMaxMax <= y[i]
+ w[i] - perm[i,k] * habitat[i] + habitat[i] + MArea*(1-iamx[i]);
/* NEW */

s.t. HabBelow{i in I}:  w[i] = sum{ k in O: (i,k) in Options}
q[i,k];
s.t. HabQ{ i in I, k in O: (i,k) in Options}:  q[i,k] <=  sum{m
in I: (i,m) in Downstream}(sum{j in I: (m,j) in Upstream}(
perm[i,k] * y[j])) - z[i,k] + sum{m in I: (i,m) in
Downstream}(perm[i,k] * habitat[m]) +sum{m in I: (i,m) in
Downstream}(perm[i,k] * w[m]);  /* NEW */

s.t. ChooseMx:  sum{i in I} iamx[i]=1;

solve;
printf {i in I: (iamx[i] !=0) }: "   The central node:  %13s
%11s     %12g  \n", i;
printf "         Barrier        Option     \n";
printf {i in I,  k in O: ((i,k) in Options) and (x[i,k] !=0) }:
"%13s   %11s     %12g  \n", i,  k,  x[i,k];

table res1{i in I,  k in O: ((i,k) in Options) and (x[i,k] !=0) }
OUT "CSV" "C:\GunnsModel_REPLACE\Res1_undirected.csv":
i~Barrier,k~Option, x[i,k]~OptionChioce;

printf "                \n";
printf "          Budget        Habitat     \n";
```

```
printf "   %12g    %12g  \n", Budget, y[FirstNod];

printf "Habitat     \n" >
"C:\GunnsModel_REPLACE\UNDIROutput.txt";              /* NEW */
printf AMaxMax >> "C:\GunnsModel_REPLACE\UNDIROutput.txt";
/* NEW */
printf  "\n The central node:     \n" >>
"C:\GunnsModel_REPLACE\UNDIROutput.txt";   /* NEW */
printf {i in I: (iamx[i] !=0) }: i >>
"C:\GunnsModel_REPLACE\UNDIROutput.txt";  /* NEW */

printf {i in I: (y[i] !=0) }: "   Y[i]     %13s     %12g  \n", i,
y[i];
table res3{i in I: (y[i] !=0) } OUT "CSV"
"C:\GunnsModel_REPLACE\Res3_undirected.csv": i~Barrier,
y[i]~Habitat;

printf {i in I,  k in O: ((i,k) in Options) and (z[i,k] !=0) }: "
z[i,k]    %13s   %11s     %12g  \n", i,  k,  z[i,k];
printf {i in I,  k in O: ((i,k) in Options) and (q[i,k] !=0) }: "
q[i,k]    %13s   %11s     %12g  \n", i,  k,  q[i,k];
printf {i in I: (iamx[i] !=0) }: "   The central node:  %13s
%11s     %12g  \n", i;
printf "The budget used:  %13s   %11s     %12g  \n", sum { i in
I,  k in O: (i,k) in Options}(cost[i,k]* x[i,k]);
printf "The maximal subnetwork:  %13s   %11s     %12g  \n",
AMaxMax;

table res2{i in I,  k in O: ((i,k) in Options) and (z[i,k] !=0) }
OUT "CSV" "C:\GunnsModel_REPLACE\Res2_undirected.csv":
i~Barrier,k~Option, z[i,k]~Habitat;

end;
```

APPENDIX B: SUPPLEMENTAL MATERIAL ON GIS PROCESSING

B.1 Troubleshooting GIS Network Creation

Troubleshooting: Barriers not Snapped / Connected

In most cases, barriers were not snapped to lines or connected to the network as expected during network creation. This problem was often detected using the ‘find disconnected’ tool in the Utility Network Analyst toolbox (ESRI, 2012b). To solve this problem, an ArcMap 'editing session' was started, all unconnected barriers selected using the ‘find disconnected’ tool, and then the ‘connect’ button was clicked on the ‘Geometric Network Editing’ toolbar. In some cases, a custom snapping tool was used, bundled with FIPEX.

Troubleshooting: Duplicate Points at Culvert or Barrier Locations

Duplicate points were found at some culverts, especially those where multiple culverts were installed.In these cases, all but one culvert point was deleted. This was usually detectable with the ‘find disconnected’ tool in the Utility Network Analyst toolbar; if points appeared disconnected after the process outlined in the ‘barriers not snapped’ section, then these points were inspected with the ‘info’ tool and duplicates deleted. Alternatively, the attribute table for the river lines was inspected and any extremely short lines were visited. Extremely short lines were usually caused by multiple barriers occurring in quick succession. Often this type of problem was associated with a network build error of 'type 11', associated with features with invalid geometry.

Troubleshooting: Line Segments Disconnected from the Network

Certain line segments were not connected to the network and could not be connected for an unknown reason. In these instances, a multistep process was needed to re-digitize the lines. First, each problematic line segment was exported as a Shapefile (i.e., duplicated and exported). Then the original was deleted and the line were re-created, tracing over the Shapefile.Numerous alternatives were attempted (e.g., ‘loading’ and ‘merging’ methods) but with no success. Ultimately, the simplest way found without needing to rebuild the network was to re-digitize these lines, a time-consuming process.

Troubleshooting: Dealing with Inter-Basin Transfers

When inter-basin transfers were encountered in the network (e.g., two watersheds were connected at headwaters), they were dealt with on a case-by-case basis. An inter-basin transfer can occur, for example, when a lake drains into two separatewatersheds. This occurred in the case of Mersey / Jordan systems at Jordan Lake and in the St. Margaret's Bay system south of Clements Lake. In the Mersey case, the network features in the Jordan system were deleted from the dataset. In the St. Margaret's Bay system, a channelization project at the point of inter-basin transfer between the east and west of the system was considered.

Troubleshooting: Braided Sections of River

Rivers at a broad scale appear to conform to a tree-like or dendritic structure. However, meso- and micro-scale deviations are common. For example, an island in a river will have two possible routes past it. This breaks the hierarchical topology governing DENs which dictates that segments may branch in an upstream direction but may not converge. A consequence of this topology is that any two points on a dendritic network have only one possible route connecting them, which is the principle on which the flow direction algorithm in the ArcGIS geometric network model is built (ESRI, 2012b). Thus, a main consequence of having looped or braided sections of river is that the geometric network is not able to determine flow direction for these sections of network. To adapt to cases where a loop in the network is unavoidable, a 'trace indeterminate flow direction'option is present in Utility Network Analyst. However, it is undesirable for large sections of river network to have indeterminate flow direction as this is problematic for identifying upstream and downstream barriers, so manual editing was sometimes necessary. In these cases, flow direction was obvious and so a small stream segment was usually 'disconnected' to break network flow in these loops and restore dendritic structure.

Network lines were edited to avoid large sections of indeterminate flow direction (Figure 178). In braided sections with barriers, network flow was always forced through only one route by disabling alternative routes. This was done in consultation with NSPI when necessary to determine the main cha nnel of passage.

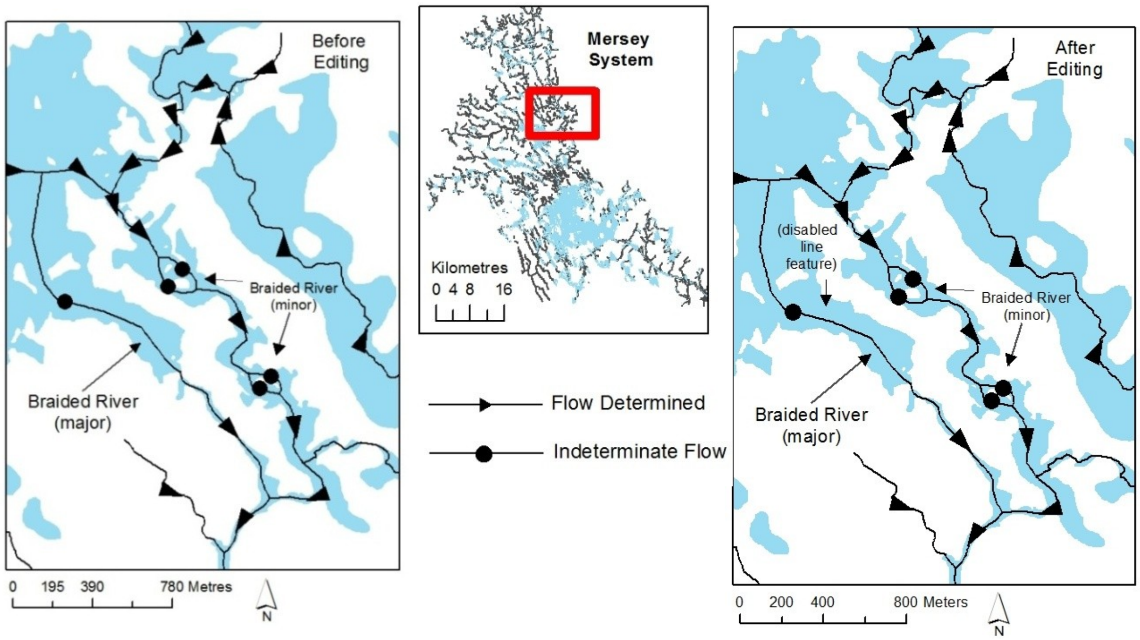

Figure B.1: Major and minor braiding in network dendritic structure. Minor braids were ignored during the network editing process whereas major ones (larger spatial extent) were not, with connectivity broken to force flow through one of the braids. If a barrier was found on a braid, they were dealt with on a case-by-case basis, and flow forced through the barrier route if necessary.

B.2 Selecting River Network for Stream Width Measurement

To determine which network lines required width estimation, the NSTDB water polygon dataset (GeoNova, 2012) was intersected with the river network line feature dataset.If network lines passed through water polygon features included in the analysis (i.e., lake, reservoir, river-lake, coastal river, or river) then the area of the polygon was used in analysis and stream widths for network lines passing through these features were not used. By specification, each network line is broken when it crosses a water polygon (Government of Nova Scotia, 2007) although this was not always the case. Steps were therefore taken to ensure lines that were not broken at the edge of a polygon were inspected and fixed in ArcGIS ArcMap Desktop 10 (Figures B.2 & B.3).

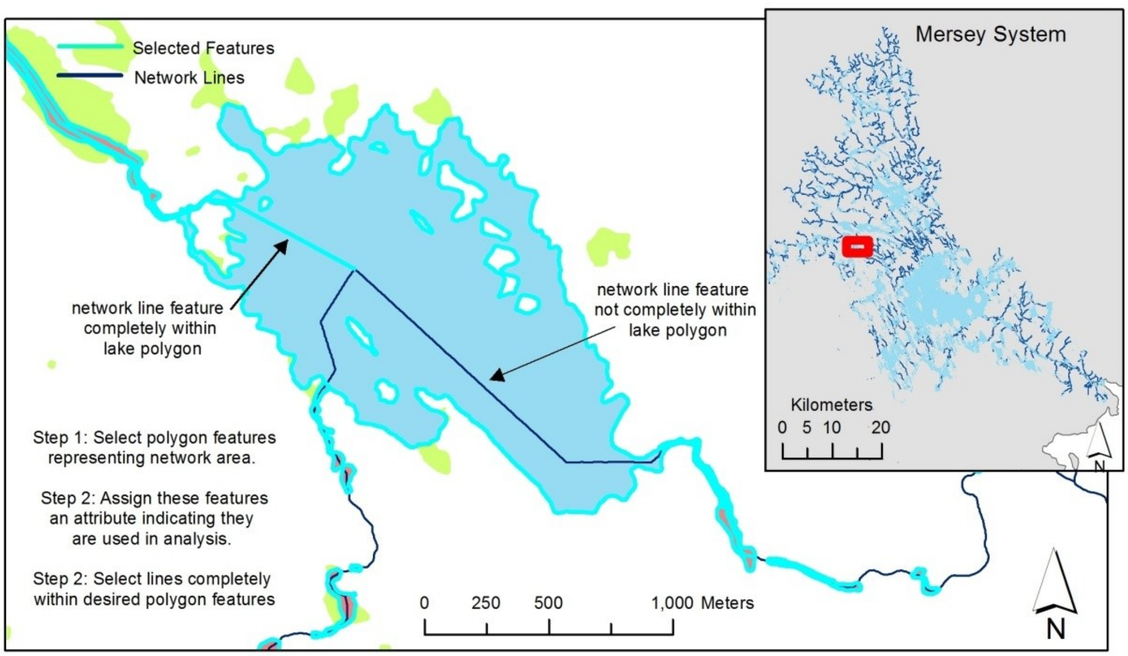

Figure B.2: Steps 1-3 of selecting network line features in need of width estimates, involving selecting all network lines completely contained within a polygon.

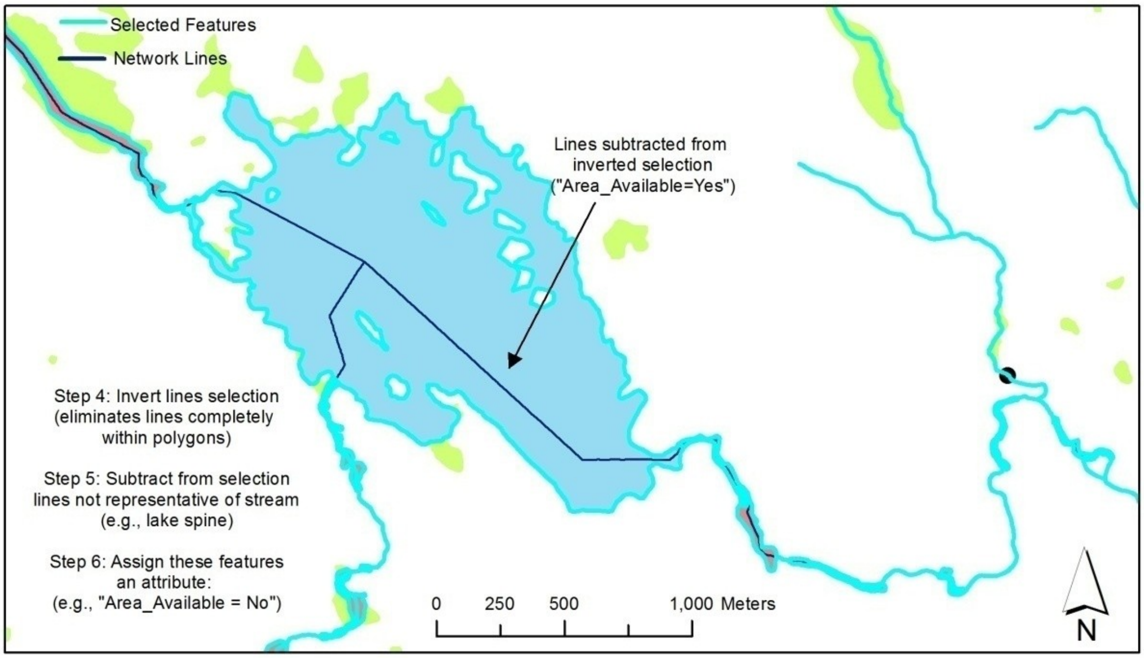

Figure B.3: Steps 4-6 of selecting network lines features in need of width estimates, involving assigning attributes to all lines outside of polygons ('Area_Available = No') and manually checking for lines features that were partially within polygons.

The following general process was used to assign an attribute ('StreamAreaAvailable'; binary values) to line features to identify those features for which no water polygon was available:

1. Used 'Select by Attributes' on river polygons for features:
   - Lake Water Area (feature code: WALK40)
   - Coastal Water Area (feature code WACO40)
   - Coast River Water Area (feature code: WACORV40)
   - Rapids (feature code: WARA40)
   - Reservoir Area (feature code: WARS40)
   - River Water Area (feature code: WARV40)
   - River Lake Water Area (feature code: WARVLK40)

   (GNS, 2007)
2. Assigned attribute to selected polygons ("UsingArea= Yes")
3. Used 'Select by Location' (setting: 'completely within' selected polygons, using existing selection from step 1) on river lines
4. Inverted selection (eliminating lines completely within polygons).

5. Subtracted from selection lines with network feature codes that indicate they are 'spine' features (eliminated features partially within water polygons and those running through a water polygon):
   - Lake spine (feature code: WALK59)
   - Coast river spine (feature code: WACORV59)
   - River lake spine (feature code: WARVLK59)

   (GNS, 2007)

   Note that in other watersheds, additional features may need to be excluded (e.g., ‘Reservoir Spine’; feature code: WRS59); however, they were not present in the selection of network lines from these three systems.
6. Assigned remaining selected line features an attribute indicating no network area available for them (‘StreamAreaAvilable = No’). Inverted selection and assign attribute indicating areas are available for them (‘StreamAreaAvailable = Yes’).
7. The selection was cleared and a selection based on ‘StreamAreaAvailable = No’ was done. Each system was thoroughly examined to identify errors, or stream lines that partially passed through water polygons. In any were found the river lines were split at the point they crossed the polygon and the appropriate ‘StreamAreaAvailable’ attribute was assigned to each resulting line.

Prior to model calibration, the relationship between stream width and other variables was explored using known widths provided by NSPI Environmental Services spatial database that included a combination of 'wetted' widths and simple width measures. Seventy-nine of these points that fell within the three watersheds of interest were used, while the others were discarded. An additional sample of 160 stream width measures was derived by examining the existing geospatial water polygon layer (‘WA_Poly’) from the NSTDB. This layer is believed to have been digitized from a mix of 1:10,000 aerial surveys and satellite imagery (GeoNova, 2012). Effort was made to sample stream and river widths from a variable and representative sample of rivers throughout each of the three river systems from headwaters to sink (n=160). River network lines that passed through water polygon features designated as ‘river’ (feature code = ‘WARV40’) were identified and river widths recorded using the 'measure' tool within ArcGIS ArcMap Desktop.