Optimising river infrastructure placement and mitigation decisions

By

Christina Ioannidou

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To Antonis and Lilian

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Chapter 1

Introduction

1.1 Background

Global freshwater biodiversity is declining rapidly (Sala et al., 2000). Habitat loss and fragmentation have severe negative effects on fish population productivity and on the integrity of aquatic ecosystems (Lucas and Baras, 2001; O'Hanley and Tomberlin, 2005). Habitat connectivity is considered critical for biological conservation (Lucas and Baras, 2001; Fahrig, 2003; Fischer and Lindenmayer, 2007) and especially for migratory fish populations, who are very vulnerable in the impediment of their movements and disruption of their life cycle (Kemp and O'Hanley, 2010; Neeson et al., 2015). Freshwater systems worldwide are heavily impacted by structures that impede the free movement of fish to essential rearing and spawning habitats affecting negatively, among others, fish population abundance and distribution (Bednarek, 2001; Lucas and Baras, 2001; Kemp and O'Hanley, 2010). Restoration of habitat connectivity with the mitigation or removal of fish passage barriers is considered a key component in the improvement of the aquatic ecosystem status (Roni et al., 2002; O'Hanley and Tomberlin, 2005).

At the same time efforts to reduce greenhouse gas emissions and address the issue of global warming have resulted in an increased interest in green energy production. Renewables are established as a mainstream source of energy worldwide (REN21, 2016). In 2015, 23.7% of the global electricity was produced by renewable sources, with hydropower providing around 16.6% (REN21, 2016). Hydropower has a well developed technology, which has been improved and refined over many years and is considered a very reliable choice for providing steady and secure power generation (ESHA, 2012). Small hydropower in particular, defined by an installed capacity of up to 10 megawatts, is a very popular option especially across Europe. Small hydropower plants (SHP) also secure water supply and flood control supporting community's needs against climate change effects (ESHA, 2012).

This thesis addresses key issues relevant to river infrastructure placement and mitigation decisions that have not been previously examined. Initially, new insights are provided regarding SHP location modeling. Hydrological issues, interactions of hydropower dams and river connectivity are all incorporated in an optimisation framework aiming to optimise SHP location strategies. Secondly, fish population and dispersal dynamics are considered in the prioritisation of barrier mitigation decisions. Spatially explicit population viability analysis (PVA) is incorporated in an optimisation framework improving the viability of migratory fish populations. Finally, a novel optimisation framework is proposed to deal with the uncertainty related to the existence of unknown barriers in river restoration planning. The effects of unknown barriers both on passability and accessible habitat are considered. The thesis consists of three papers that are presented in the next chapters. A review of the relevant literature is presented in the section below.

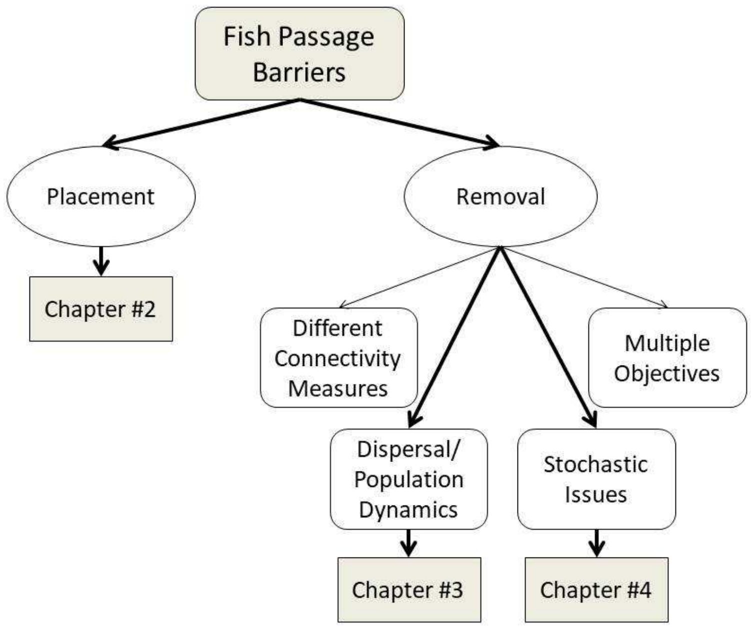
1.2 Literature Review

River infrastructure serves many of society's needs, e.g., transportation, flood control, power production, but also disrupts natural river continuity and severely affects aquatic community's composition (Doyle and Havlick, 2009). In stream structures that impede fish movement vary significantly in type (e.g., culverts, road crossings, hydropower dams), can be either full or partial barriers, and also have varying effects ranging from short delays to complete blockage of fish passage (Kemp and O'Hanley, 2010). The effects that fish passage barriers have on the aquatic continuity have been widely studied and several approaches have been suggested so that river connectivity is taken into account in the water management planning. The relevant literature can be divided into two main categories: the studies that address the issue of placing new barriers (e.g., hydropower dams) and the studies that deal with the removal or mitigation of already existing barriers. An overview of the literature relevant to fish passage barriers is shown in Figure 1.1.

1.2.1 Placement of Hydropower Plants

The literature on placement of new barriers is relatively limited and is restricted to hydropower dam location. The increasing availability of satellite imagery and the rapid development of remote sensing technologies have allowed the extraction of various topographic and hydrologic characteristics, critical in determining the suitability of sites. Geographic information systems (GIS) are widely used in spotting feasible sites and assessing their power generation potential (Coskun et al., 2010; Cyr et al., 2011; Dudhani et al., 2006; Kusre

Figure 1.1: An overview of the literature relevant to fish passage barriers.



et al., 2010; Ramachandra et al., 2004). While most studies that focus on searching for candidate hydropower sites do not take into consideration any environmental issues, there are some exceptions where environmental criteria are taken into account (Lee et al., 2008; Rojanamon et al., 2009; Yi et al., 2010).

Installation decisions are considered independently in almost every proposed methodology. An exception is Larentis et al. (2010), where the interactive effects of hydropower dams are considered. The proposed methodology treats the hydropower installations in a basin as a system, where the installation of a hydropower plant influences the power generation of the downstream sites as it increases the water discharge. Kusre et al. (2010) also considered potential interactions between hydropower dams. In particular, they set a minimum distance between any two consecutive plants in order to avoid any reductions on the generation potential of upstream sites by the raising of the water surface profile by the dam downstream (known as "backwater effect").

Another relevant study is the one carried out by the UK's Environment Agency (EA). EA, in an effort to comply with EU demands to make the most out of all available renewable resources, conducted a survey on small scale hydropower potential in England and Wales (Agency, 2010). All known weirs were considered as possible hydropower plant locations and a variety of methods was used to estimate the flow and hydraulic head values. All candidate sites were categorized based on their power generation potential and their environmental sensitivities (i.e., presence of key fish species or areas of special conservation concern). A very interesting approach is the study by Ziv et al. (2012). Rather than employ a typical GIS assessment of site feasibility, they develop a framework that incorporates spatially-explicit fish dispersal and population growth models to investigate in detail the ecological impacts of hydropower development. They explore the trade-offs between hydropower, fish biomass, and biodiversity within the Mekong River Basin. The authors analyzed all possible dam development scenarios, which limits the scalability of their approach to problems involving small numbers of possible dam locations. To our knowledge, the only existing example in the literature that uses optimisation techniques in the planning of hydropower development is the study of Chang et al. (1992). The authors aim to maximize hydropower production while quantifying the trade-offs between power generation and water quality in terms of dissolved oxygen concentrations. A case study of the upper Ohio river basin illustrates the benefits of using of the proposed framework.

The first paper presented in this thesis, titled "Eco-friendly location of small hydropower", proposes a formal optimisation framework for locating small hydropower dams in an environmentally friendly way. A multi-objective optimisation model is used to maximize total hydropower production while considering the overall river connectivity. The non-linear initial form of the optimisation model is linearized through a series of steps to a mixed integer linear programming model. In our analysis we take into account the "backwater effects" that the installation of hydropower plants has on the water surface profiles upstream affecting both the hydropower generation potential of the nearby sites and the capability of fish to pass successfully these sites. A case study in England and Wales is used to illustrate the usefulness of the proposed framework. Interestingly, according to our findings, in river networks heavily disrupted by fish barriers the installation of small hydropower plants fitted with fish passes may actually create a win-win situation where maximizing hydropower production also improves river connectivity.

1.2.2 Fish Passage Barrier Mitigation

The other two papers presented in this thesis fall under the second general category of the barrier related literature, that deals with the issue of barrier removal. Barrier removal is widely studied and many methodologies have been suggested to prioritize barrier mitigation decisions. The techniques that are being used vary from simple scoring and ranking approaches (e.g., Karle, 2005; Kocovsky et al., 2009; Nunn and Cowx, 2012) to far more sophisticated optimisation methods (e.g., Paulsen and Wernstedt 1995; O'Hanley and Tomberlin 2005; Kuby et al. 2005; Zheng et al. 2009; O'Hanley et al. 2013; King and O'Hanley 2016). Scoring and ranking techniques are easy to implement but cannot capture the spatial arrangement of barriers in the river network. The interactive effects that barrier removal decisions hold are not considered which can result to highly inefficient solutions (O'Hanley and Tomberlin, 2005). On the other hand, optimisation methods can treat the whole river network as a system, where barriers are interconnected and each mitigation decision can change the dynamics of the whole system (Kemp and O'Hanley, 2010). As pointed out in Kemp and O'Hanley (2010) optimisation techniques are the ideal option as they can guarantee to maximize the restoration gains while considering the operational and resource restrictions.

1.2.2.1 Connectivity Measures

The most common connectivity measure considered in optimisation based studies is the amount of accessible (i.e., connectivity weighted) habitat available to migratory fish. For example, O'Hanley and Tomberlin (2005) devised a nonlinear integer program to maximize the total net gain in accessible habitat by optimizing the removal or repair of full and partial fish passage barriers subject to a budgetary constraint. Their optimisation approach manages to capture the spatial barrier network and the interactive effects that barrier removal decisions have on the overall river connectivity. Similarly, Kuby et al. (2005) use a bi-objective optimisation model aiming to maximize the amount of reconnected salmon habitat by the removal of large hydropower dams. O'Hanley (2011) presents an optimisation model for prioritizing the removal of artificial passage and flow barriers in order to restore free-flowing conditions over the widest extent possible. The objective is to decide which barriers to remove, given a limited budget, in order to maximize the length of the single largest unimpeded subsection of the river. The studies by Kuby et al. (2005) and O'Hanley (2011) assume that artificial barriers are either fully passable or not. This assumption is useful in the sense that model's data requirements are minimal especially when reliable passability values cannot be obtained.

In the cases where passability data are available more sophisticated connectivity measures can be devised. For example, Cote et al. (2009) introduce two metrics to describe the longitudinal connectivity of river networks for potadromous and diadromous fish. Their index reflects the probability that fish can move between any two randomly chosen points in a river network. Their analysis is based on carrying out complete enumeration of all possible barrier removal scenarios limiting the scalability of their approach to cases where there are relatively few barriers for removal. Diebel et al. (2010) develop a new metric (C) to quantify stream connectivity for resident fish. Their metric accounts for the amount, quality, and level of connectivity to different stream habitat types and also considers the accessibility that stream-resident fish have on the different river segments. A greedy type heuristic is used to rank barriers for mitigation. O'Hanley et al. (2013) extend the work of Diebel et al. (2010) by developing an optimisation model for prioritizing the removal of resident fish passage barriers. Their goal is to maximize longitudinal connectivity for stream resident fish and other aquatic species, measured by the C metric, properly modified to account for multiple watersheds, given a limited budget. The original nonlinear model is reformulated as an exact mixed integer linear program.

1.2.2.2 Fish Dispersal and Population Dynamics

There are a few optimisation studies that take into account fish dispersal and population dynamics in the river restoration planning. An example is the work by Paulsen and Wernstedt (1995) where the authors develop a linear programming model to identify the least cost solution for restoring salmon populations affected by hydropower dams in the Columbia River basin. The effects of all possible management alternatives on the population dynamics were evaluated by the use of deterministic, life-cycle-simulation models. The lengthy simulation runs limit the scalability of the suggested approach (Kemp and O'Hanley, 2010). Zheng et al. (2009) and later Zheng and Hobbs (2013) also considered fish population health using a rather simplistic approach where 8 criteria are aggregated in a fish health measure.

In another relevant study Newbold and Siikamaki (2009) present a framework to prioritize watershed conservation activities. In their approach they develop a population viability analysis (PVA) model to estimate the long-run probability of persistence for salmon and a habitat quality model. A reserve site selection procedure combines the two models and prioritizes the watershed protection decisions based on their cost-effectiveness.

Looking beyond optimisation based studies, PVA models have seen much wider use in the context of river habitat. An example is the study of Harvey and Railsback (2012) where the authors explore the effects of barriers on a virtual stream trout population. They use a detailed individual-based model which captures the mechanisms by which habitat dynamics and individual fish behavior affect movement and population growth of the subpopulations separated by barriers. Five scenarios with varying barrier densities were simulated to investigate how the location of barriers affects two population stability properties: persistence and resistance. Interestingly, according to their findings, low barrier densities can actually increase biomass.

Another example is the study by Nieland et al. (2015). They developed a dam impact analysis model to evaluate the demographic effects of dams on migratory fish populations. Natural and dam-related mortality along with the numbers and locations of fish at multiple life stages were incorporated into a multi-state PVA. They simulated the effects of 6 different removal scenarios on the mortality of Atlantic salmon populations in the Penobscot River in Maine. Their analysis aims to assess the responses of fish populations to the different barrier removal scenarios rather than to predict absolute abundance.

Nickelson and Lawson (1998) develop a life cycle model based on habitat quality to estimate the extinction risk for coho salmon populations along the Oregon Coast. Their model accounts for environmental, demographic and genetic stochasticity and also considers fish straying. The long term viability of coho salmon is evaluated by 99-year simulations. Scheuerell et al. (2006) developed a framework to assess the salmon population responses to changes in habitat, hatchery operations, and harvest levels. Their approach relies on a multistage

Beverton-Holt population model, considers relationships among habitat attributes, fish survival, and carrying capacity and provides estimates on abundance, productivity, spatial structure, and diversity. Sweka and Wainwright (2014) provide a review on the use of PVA models in the planning of recovery actions for Atlantic and Pacific salmon.

In the second paper presented in this thesis, titled: "The importance of spatiotemporal fish population dynamics in barrier mitigation planning", spatially explicit PVA is incorporated in a formal optimisation framework to prioritize barrier mitigation decisions aiming to improve the viability of migratory fish populations. The population and dispersal dynamics of a wild coho salmon population from the Tillamook basin along the Oregon Coast of the USA are explored in a case study. Two extreme homing patterns are considered, river versus reach homing, density dependence is assumed and fish straying is taken into account. According to our findings barrier removal decisions are highly affected by the level of homing fidelity. With reach homing almost the same barrier removal scenarios maximize accessible habitat and fish population numbers. With river homing, on the other hand, maximum population size can be reached without removing all the barriers. A stochastic version of our model reveals that removing all the barriers actually results in a marginal increase of quasi-extinction probability.

1.2.2.3 Stochastic Issues

Studies relevant to barrier removal prioritization rarely address any type of uncertainty. An exception is the study of McKay et al. (2013). The authors use a graph-theoretic approach to prioritize barrier improvement decisions. In order to assess upstream fish passage connectivity they introduce a habitat connectivity index that accounts for uncertainty in barrier passability values. Their model was applied to prioritize restoration activities in Truckee River in Nevada, USA.

The third paper presented in this thesis, titled: "The hidden elephant in the room: Large-scale river connectivity restoration requires planning for the presence of unrecorded barriers" introduces a novel optimisation framework to prioritize barrier mitigation decisions while accounting for the uncertainty related to the existence of unknown or "hidden" barriers. Barrier datasets are never exhaustive in recording all the actual obstacles blocking fish passage but this reality has been ignored so far by all relevant studies. Our approach considers the effects that the existence of "hidden" barriers has on the effective accessible habitat and on the cumulative passability values. By applying our framework in a case study in the state of Maine in USA we find that there is a dramatic decrease in longitudinal connectivity gains even if a small percentage of hidden barriers are present. Taking into account hidden barrier uncertainty into the optimisation process substantially improves the potential gains in accessible habitat. Anticipating for hidden barriers results to far more effective river restoration planning.

1.2.2.4 Multiple Objectives

Multi-objective optimisation is an approach capable of dealing with the multiple environmental, economic and social goals and constraints, often conflicting, that river restoration planning involves (Kemp and O'Hanley, 2010). For example, Kuby et al. (2005) developed a multi-objective optimisation framework for prioritizing the removal of large hydropower dams. Their model captures the trade-offs between ecological gains for migratory fish and economic losses by systematically incorporating river connectivity into the decision making process. Their model was used to identify alternative dam removal scenarios in the Willamette river watershed in Oregon, USA.

In another relevant study Zheng et al. (2009) propose the use of a multi-objective optimisation model for the habitat restoration in the Lake Erie basin. The model quantifies the trade-offs between the ecological (e.g., native species biomass), socio-economic (e.g., recreational and commercial harvesting) and economic (e.g., dam removal cost) goals. Alternative dam removal scenarios are identified that vary in terms of their ecological and socio-economic benefits. Zheng and Hobbs (2013) extend the work of Zheng et al. (2009) by considering an additional goal of reducing the risk of dam failure.

1.3 Outline

The rest of this thesis is organised as follows. In Chapter 2 a formal optimisation framework for locating small hydropower dams in an environmentally friendly way is proposed. In Chapter 3 spatially explicit PVA is incorporated in an optimisation framework to prioritise barrier mitigation decisions aiming to improve the viability of migratory fish populations. In Chapter 4 a novel optimisation framework to prioritize barrier mitigation decisions while accounting for the uncertainty related to the existence of unknown barriers is introduced. Finally, the conclusions are discussed in Chapter 5.

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Chapter 2

Eco-Friendly Location of Small Hydropower

We address the problem of locating small hydropower dams in an environmentally friendly manner. We propose the use of a multi-objective optimization model to maximize total hydropower production, while limiting negative impacts on river connectivity. Critically, we consider the so called "backwater effects" that dams have on power generation at nearby upstream sites via changes in water surface profiles. We further account for the likelihood that migratory fish and other aquatic species can successfully pass hydropower dams and other artificial/natural barriers and how this is influenced by backwater effects. Although naturally represented in nonlinear form, we manage through a series of linearization steps to formulate a mixed integer linear programing model. We illustrate the utility of our proposed framework using a case study from England and Wales. Interestingly, we show that for England and Wales, a region heavily impacted by a large number of existing river barriers, installation of small hydropower dams fitted with even moderately effective fish passes can, in fact, create a win-win situation that results in increased hydropower and improved river connectivity.

2.1 Introduction

Efforts to reduce carbon emissions in both industrialized and developing countries has resulted in an increased interest in renewable energy production. Hydropower, in particular, has gained special attention. Although installation costs can be appreciable, operating costs are generally low, the technology is already well developed, and of the many other sources of renewable energy (e.g., wind and solar) it is far more reliable in terms of providing base load power generation. Among the various types, small hydropower plants (SHP) with an installed capacity of up to 10 MW are by far the most common and logistically feasible option in many places, particularly across Europe. According to the European Small Hydropower Association, SHP currently supplies enough electricity for 13 million households and plays a key role in greenhouse gas (GHG) emissions reduction through green energy production (ESHA, 2012). It also supports water management policies, aids in climate change adaptation through flood control, and contributes to the prevention of water scarcity and drought.

In the UK, the government has set a goal of reducing emissions by 18% by 2020 (HM Government, 2009a). Renewable energy is considered a key part of the overall plan with respect to electricity generation. In particular, the UK Renewable Energy Strategy has set a legally-binding target to ensure that 15% of energy production comes from renewable sources by 2020 (HM Government, 2009b). Even if small-scale hydropower is not expected to play a major role in this, the ambition is such that all sources of renewable energy are expected to deliver their maximum sustainable potential HM Government (2009a). In particular, according to the UK's National Renewable Energy Action Plan (DECC, 2010), new SHP schemes of between 40 MW and 50 MW need to be installed annually until 2020.

Although clean in terms of GHG emissions, the installation of hydropower schemes can nonetheless have adverse impact on the local environment, especially on fish populations and other aspects of river ecosystems (Stanford et al., 1996; Bednarek, 2001; Roni et al., 2002; O'Hanley and Tomberlin, 2005). Hydropower dams form physical barriers that often disrupt the natural connectivity of rivers by reducing water and sediment transfer, which can impact geomorphology processes and fragment river habitats. In particular, dams can impede fish access to essential breeding and rearing areas, resulting in reduced fish productivity and other changes in aquatic community composition (Lucas and Baras, 2001). Hence, any decision about installing hydropower dams normally involves a trade-off between renewable energy production on the one hand and healthy rivers on the other. This highlights the need for decision support tools in SHP location planning, which are capable of balancing these two basic but competing goals. Such tools would prove extremely useful to river management organizations in devising more sound and effective hydropower development strategies.

In this chapter, the problem of optimally locating SHPs is addressed. We propose a series of integer programing models for siting SHPs in order to maximize overall hydropower generation capacity while limiting negative impacts on river connectivity. Studies thus far have focused on searching for a set of feasible locations for installing SHP rather than optimizing site selection.

2.1.1 Hydropower Location

Much of the literature on hydropower location focuses on the use of geographic information systems (GIS) to screen for potential dam locations, driven in large part by the increasing availability of satellite imagery and other remotely sensed data. Site feasibility and power generation potential are usually the two main concerns (Ramachandra et al., 2004; Dudhani et al., 2006; Coskun et al., 2010; Kusre et al., 2010; Cyr et al., 2011), with only occasional treatment of environmental aspects (Lee et al., 2008; Rojanamon et al., 2009; Yi et al., 2010). A good example is the study by Yi et al. (2010), which uses a combination of hydrologic, topographic, and environmental criteria to rate the suitability of candidate SHP sites. Using a case study area in South Korea, a small set of promising locations for reservoir and run-of-river type SHPs is identified by performing a series of geospatial data processing steps.

Installation decisions are considered independently in almost every proposed methodology. An exception is Larentis et al. (2010), where the interactive effects of hydropower dams are considered. The proposed methodology treats total hydropower in a subbasin as a system, where the siting of a dam reduces the generation potential of upstream sites by raising the water surface depth (the so called "backwater" effect explained in more detail in Section 2.2.3). Maximum hydropower potential within a basin is estimated by siting dams in series along a river course, such that each dam lies outside the length of the backwater curve produced by the dam downstream.

Of particular relevance to our current work is the study by Ziv et al. (2012). Rather than employ a typical GIS approach, the authors examine in detail the ecological impacts of hydropower development within the Mekong River Basin. Their framework, which incorporates spatially-explicit fish dispersal and population growth models, is designed to explore trade-offs between hydropower, fish abundance, and biodiversity. Trade-off curves are produced by enumerating all possible dam development scenarios, which invariably limits the scalability of their approach to problems involving small numbers of possible dam locations.

Another relevant study is one carried out by the UK's Environment Agency (EA), which looked into the potential for expanding renewable energy production from small scale hydropower across England and Wales (EA, 2010). All known weirs were considered as possible hydropower plant locations. Using a variety of methods to estimate flow, weirs were assessed for their hydropower potential and subsequently categorized based on their environmental sensitivities (i.e., presence of key fish species or areas of special conservation concern).

To our knowledge, Chang et al. (1992) is the only existing example in the literature to propose a formal optimization framework for selecting hydropower development alternatives. Their methodology takes into account potential reductions in water quality (measured in terms of dissolved oxygen concentrations) caused by the installation of hydropower dams. Using a case study of the upper Ohio basin, they investigate trade-offs between power generation and water quality.

2.1.2 Barrier Mitigation Planning

While there are few examples involving the use of optimization techniques for locating new hydropower dams (Chang et al., 1992), optimization has been applied frequently in the context of cost-effectively removing of dams and other river infrastructure to improve river connectivity. Some examples include: Paulsen and Wernstedt (1995), O'Hanley and Tomberlin (2005), O'Hanley (2011), O'Hanley et al. (2013b), and Neeson et al. (2015). A key feature of these studies and other similar optimization based approaches is the explicit consideration of the spatial structure of barrier networks and the interactive effects that barrier removal decisions have on longitudinal connectivity.

One study dealing specifically with hydropower is Kuby et al. (2005), who propose the use of a multi-objective optimization model for prioritizing the removal of large hydropower dams. Their model quantifies trade-offs between ecological gains for migratory fish, economic losses from reduced hydropower generation and water storage capacity. The use of a multi-objective framework is noteworthy in that it offers decision makers a means of identifying alternative portfolios of dam removal that vary in terms of their ecological and socioeconomic benefits. This, in turn, can help to inform negotiations among managers and different stakeholders.

Zheng et al. (2009) propose a mixed integer linear programing model for optimizing the net benefits of removing multiple dams in the Lake Erie basin. The model is multi-objective and aims to maximize a combination of ecological (e.g., native species biomass) and socio-economic (e.g., recreational and commercial harvesting) goals subject to a budget constraint. Zheng and Hobbs (2013) extend the model proposed by Zheng et al. (2009) by adding the additional goal of reducing the risk of dam failure.

A detailed review of procedures and techniques related to evaluating and prioritizing the mitigation of fish passage barriers can be found in Kemp and O'Hanley (2010). Given multiple and often conflicting environmental and economic goals, they recommend the use of optimization models and multi-criteria decision making techniques as an objective and efficient means for prioritizing barrier repair and removal decisions.

The remainder of the Chapter is organized as follows. In Section 2.2, we present the hydropower plant location problem. Specifically, in Section 2.2.1, we present a basic nonlinear model and in Section 2.2.2 a linear reformulation. In Section 2.2.3, we talk briefly about the backwater effect caused by siting a dam.

This is followed in Section 2.2.4 by the development of an extended version of the hydropower plant location problem, where backwater effects are considered. In Section 2.3, we apply our methodology to a case study of England and Wales and discuss key findings. Finally, in Section 2.4, we give some concluding remarks.

2.2 Hydropower Plant Location Problem

The aim of the hydropower plant location problem (HPLP) is to select sites for installing dams to maximize potential hydropower generation while keeping longitudinal river connectivity at or above some specified lower bound. Given a range of dam sizing options for each potential dam location, the hydropower potential w_{ji} (measured in Watts) at site j when fitted with a dam of size i is defined by the well-known equation:

$$w_{ji} = \eta_{ji} \rho g Q_j H_{ji} \tag{2.1}$$

where η_{ji} is the efficiency (in the range 0-1) of the dam's turbine, ρ is the density of water (1000 kg/m^3), g is the acceleration due to gravity (9.81 m/s^2), Q_j is the river's volumetric flow (m^3/s) at site j, and H_{ji} is the hydraulic head (m) of a dam with size i (i.e., the difference in water surface height above and below the dam).

Hydropower dams and other artificial or natural barriers that may be present within a river network are assumed to allow partial fish passage. More formally, the passability of a barrier refers to the fraction of fish, in the range $[0, 1]$, that are able to successfully navigate it in the upstream and or downstream direction, where 0 denotes a completely impassable structure and 1 a completely passable one (Kemp and O'Hanley, 2010). Typically, barriers with larger head heights are more difficult to pass as fish need to leap higher. Cumulative passability, which is synonymous with longitudinal connectivity, describes the collective impact that multiple barriers have on fish dispersal. Assuming barrier passabilities are independent, cumulative passability to an area immediately above any barrier is evaluated by multiplying the barrier's passability by the passabilities of any downstream barriers to the river mouth. To ensure that longitudinal connectivity is not excessively compromised by the installation of hydropower dams, a constraint is included in the model HPLP requiring cumulative passability weighted habitat above hydropower dams and other barriers to be greater than or equal to a user-defined threshold. For each dam sizing option, a different barrier passability value can be assigned depending on the dam's height and what options are available for constructing an effective fish pass (e.g., fish ladder, fish elevator, or bypass channel).

2.2.1 Basic Model

In order to formulate a basic version of HPLP, let N, indexed by j, be the set of candidate hydropower dam sites. For each dam site $j \in N$, set S_j, indexed by i, specifies the dam sizing options available at j. Installation of a dam of size i at site j results in a hydropower potential of w_{ji}, as determined by equation (2.1). In addition to locating new dams, other artificial and natural barriers, which invariably impact fish passage and longitudinal connectivity, may already be present in a river network. These are denoted by the set B, indexed by j, while the set J, indexed by j and k, is used to denote all existing artificial/natural barriers plus candidate dam sites (i.e., $J = N \cup B$). It is assumed throughout that a river has a strictly "dendritic" structure, meaning that it never diverges in the downstream direction, thus excluding braided river systems. In effect, this implies that 1) the set of potential barrier locations J forms a tree network with each location $j \in J$ having at most one downstream site and 2) there is a unique path from the river mouth to any upstream location.

To continue, the set $D_j \subseteq J$ specifies all potential barriers downstream from and including site $j \in J$. For each location $j \in J$, the quantity v_j denotes the net amount of habitat (measured in terms of length or area) upstream of j to the next set of potential barriers or the ends of the river network. Parameter p_j^0 refers to the current passability of site $j \in J$, while p_{ji} refers to the change in passability at site $j \in N$ when a dam of size i is built there. Note that p_{ji} can be negative (a decrease in passability), positive (an increase in passability), or zero (no change in passability) depending on what type of dam and/or fish passage structure is installed.

This requires some further explanation. In general, installation of a dam will cause a decrease in fish passage. However, in certain situations (as with our study area discussed below), it may be feasible to locate hydropower dams at existing artificial or natural barriers, which have current passabilities well below 1 (i.e., if $N \cap B \neq \emptyset$). If a dam were to be located at such a site and fitted with a suitable fish pass, then it is entirely possible for passability to increase above its current baseline.

Finally, let V^0 be the total amount of habitat currently accessible to fish (i.e., $V^0 = \sum_j P_j^0 v_j$, where $P_j^0 = \prod_{k \in D_j} p_k^0$) and let $\alpha \geq 0$ be a scaling parameter for determining the minimum amount of accessible habitat that needs to be achieved following the siting of hydropower dams.

Using the following decision variables:

$$x_{ji} = \begin{cases} 1 & \text{if a hydropower dam of size } i \text{ is installed at site } j \\ 0 & \text{otherwise} \end{cases}$$

$z_j =$ cumulative passability to river habitat immediately above location j

a nonlinear formulation for HPLP is given as follows:

$$[\text{HPLP1}]\ \max \sum_{j \in N} \sum_{i \in S_j} w_{ji} x_{ji} \tag{2.2}$$

s.t.

$$z_j = \prod_{k \in D_j} \left(p_k^0 + \sum_{i \in S_k} p_{ki} x_{ki} \right) \qquad \forall j \in J \tag{2.3}$$

$$\sum_{j \in J} v_j z_j \geq \alpha V^0 \tag{2.4}$$

$$\sum_{i \in S_j} x_{ji} \leq 1 \qquad \forall j \in N \tag{2.5}$$

$$x_{ji} \in \{0, 1\} \qquad \forall j \in N, i \in S_j \tag{2.6}$$

The objective function (2.2) maximizes the sum of hydropower potential across all candidate dam sites. The first set of constraints (2.3) calculates the cumulative passability of each site j. Cumulative passability z_j equals the product of the passability of site j and the passabilities of all downstream sites to the river mouth. The passability of site j equals initial passability p_j^0 plus any change in passability p_{ji} if a hydropower dam of size i is installed at j ($x_{ji} = 1$). Constraint (2.4) guarantees that total cumulative passability weighted habitat is bounded below by some multiple α of the current amount of accessible habitat V_0 within the study area. Constraints (2.5) guarantee that at most one hydropower sizing option is selected at site j. Finally, constraints (2.6) force the x_{ji} dam location variables to be binary.

2.2.2 Linear Reformulation

To reformulate [HPLP1] as a mixed integer linear program, we introduce the following additional variables:

y_{ji} = change in cumulative passability at site j given installation of dam size i

Variable y_{ji} equals 0 if there is no change in cumulative passability at site j and is positive/negative given an increase/decrease in cumulative passability. Further, let $d_j \in D_j$ refer to the potential barrier immediately downstream of j, if one exists. A linear version of the basic HPLP problem can be derived by replacing equations (2.3) with the following constraints:

$$z_j = \begin{cases} p_j^0 + \sum_{i \in S_j} y_{ji} & D_j = \emptyset \\ p_j^0 z_{d_j} + \sum_{i \in S_j} y_{ji} & D_j \neq \emptyset \end{cases} \qquad \forall j \in J \tag{2.7}$$

$$y_{ji} = p_{ji} x_{ji} \qquad \forall j \in N, i \in S_j | D_j = \emptyset \tag{2.8}$$

$$y_{ji} \leq p_{ji} z_{d_j} - p_{ji}\left(1 - x_{ji}\right) \qquad \forall j \in N, i \in S_j | D_j \neq \emptyset \wedge p_{ji} < 0 \tag{2.9}$$

$$y_{ji} \leq 0 \qquad \forall j \in N, i \in S_j | D_j \neq \emptyset \wedge p_{ji} < 0 \tag{2.10}$$

$$y_{ji} \leq p_{ji} x_{ji} \qquad \forall j \in N, i \in S_j | D_j \neq \emptyset \wedge p_{ji} \geq 0 \tag{2.11}$$

$$y_{ji} \leq p_{ji} z_{d_j} \qquad \forall j \in N, i \in S_j | D_j \neq \emptyset \wedge p_{ji} \geq 0 \tag{2.12}$$

The z_j and y_{ji} variables, in combination with constraints (2.7) - (2.12), form a series of "probability chains" (O'Hanley et al., 2013a) that recursively evaluate the cumulative passability of each site j based on the cumulative passability downstream from j. In particular, equations (2.7) determine the cumulative passability for each site. There are two cases. If site j has no potential downstream barrier ($D_j = \emptyset$), then cumulative is equal to the initial passability p_j^0 at j plus any change in cumulative passability $\sum_{i \in S_j} y_{ji}$ resulting from the installation of a dam at j. Alternatively, if site j does have at least one downstream site ($D_j \neq \emptyset$), then the initial passability p_j^0 at j needs to be further multiplied by the cumulative passability z_{d_j} of j's downstream site d_j.

Collectively, constraints (2.8) - (2.12) determine changes in cumulative passability y_{ji} due to dam installation. If site j has no potential downstream barrier ($D_j = \emptyset$), equations (2.8) simply state that the change in cumulative passability y_{ji} due to the installation of a dam of size i is equal to p_{ji} if a dam is located there ($x_{ji} = 1$), 0 otherwise ($x_{ji} = 0$). For sites with at least one potential downstream barrier ($D_j \neq \emptyset$), inequalities (2.9) and (2.10) apply in cases where dam installation would cause a decrease in passability ($p_{ji} < 0$), while inequalities (2.11) and (2.12) apply if dam installation would potentially cause an increase in passability ($p_{ji} \geq 0$). In either situation, they place an upper bound of $p_{ji} z_{d_j}$ on variable y_{ji} whenever a dam is located at site j ($x_{ji} = 1$), 0 otherwise.

It is worth pointing out that the upper bounds on the y_{ji} variables imposed by (2.9) - (2.12) are not guaranteed to be strictly binding. Implicitly, there is a preference for increases (decreases) in cumulative passability to be as large (small) as possible in order to satisfy the minimum accessible habitat constraint (2.4) (i.e., by having the y_{ji} variables equal to their upper bounds). However, in situations where the siting of dams produces a slack in constraint (2.4), it is possible for one or more y_{ji} variables to be less than their specified upper bounds and still satisfy constraint (2.4). While this in no way affects the optimality of the x_{ji} variables, values for the y_{ji} variables and hence total accessible habitat $\sum_{j \in J} v_j z_j$ may be incorrectly specified. To be more clear, the optimality of the x_{ji} variables is not affected by the values of the y_{ji} variables as what matters for the x_{ji} variables is that constraint (2.4) is satisfied and not what the exact value of the total accessible habitat. So, given that a solution satisfies constraint (2.4), y_{ji} variables can either reach or not reach their upper bounds without affecting the optimality or feasibility of the solution.

To determine precisely changes in cumulative passability, one can perform a simple post-processing step, after an optimal solution for the x_{ji} variables has been found, in which the y_{ji} variables for sites $j \in J$ with at least one downstream barrier ($D_j \neq \emptyset$) are iteratively set to $p_{ji} z_{d_j} x_{ji}$ starting with the most downstream sites (i.e., $|D_j| = 1$) and progressively moving in the upstream direction. Alternatively, one can include a secondary objective in an attempt to force the y_{ji} variables to their upper bounds. More specifically, this can be achieved by adding $\varepsilon \sum_{j \in N} \sum_{i \in S_j} y_{ji}$ to the objective function (2.2), where $\varepsilon > 0$ is some very small weight less than the minimum difference between any pair of hydropower potential values w_{ji} (e.g., $\varepsilon = 0.99 \times \min_{j,k \in N, i \in S_j, t \in S_k} \{w_{ji} - w_{kt}\}$). In our implementation, we used the post-processing option.

2.2.3 Backwater Effects on Hydropower Potential

In model [HPLP1], hydropower potential at each candidate site is assumed to be independent of the spatial arrangement of dams, which does not necessarily hold in reality. In particular, the presence of a dam within a watercourse (or any in-stream structure) invariably causes an increase in the water surface behind the dam, which gradually decreases as one moves in the upstream direction (Figure 2.1). This change in the water surface profile of a river is called the “backwater effect” and is described by the backwater curve, which determines, based on slope and flow characteristics, the depth of water at any given point upstream.

Backwater curves are important when evaluating the hydropower potential of sites. The presence of a dam can cause a reduction in head difference (due to increased water depth) and hence a reduction in hydropower potential at upstream sites. One option for dealing with backwater effects, akin to Kusre et al. (2010), would be to include additional constraints in [HPLP1] that prevent nearby dams from being simultaneously located if and when the change in head difference at the upstream site (caused by the presence of a dam downstream) exceeds some threshold. The alternative, details of which are given below, is to explicitly incorporate backwater effects into a more realistic but complex model.

To formulate a hydropower plant location model with interactive backwater effects, consider the following

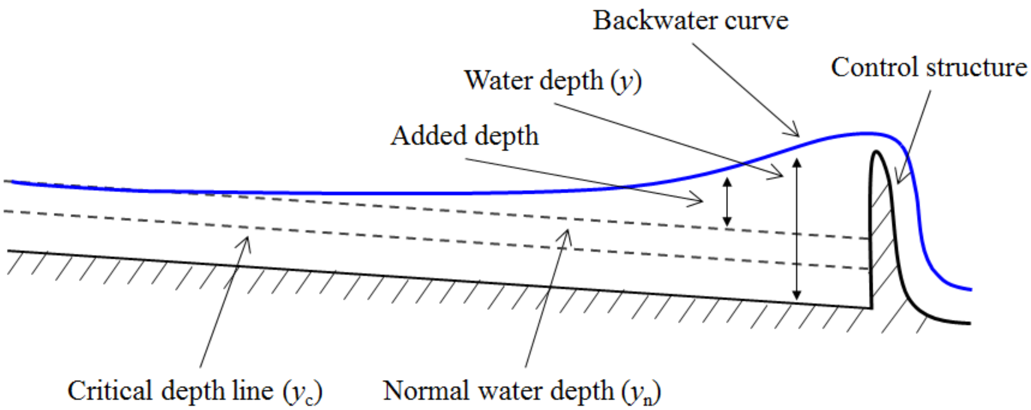


Figure 2.1: Representative backwater profile for a mild M1-type curve ($y > y_{\mathrm{n}} > y_{\mathrm{c}}$).

additional notation. Let M_j be the set of sites downstream from j that can potentially have a backwater effect on site j and let $I_{jk} = D_j \setminus (\{j\} \cup D_k)$ be the set of sites lying between j and k. Assuming no dams are located in set I_{jk} (i.e., $x_{\ell s} = 0$, $\forall \ell \in I_{jk}, s \in S_k$), the reduction in head due the backwater effect caused by a dam of size t located at downstream site $k \in M_j$ is denoted by ΔH_{jkt}. In other words, the actual head height at site j is the difference between the original head height of the site, H_{ji}, minus any potential increase of the water surface profile at j, ΔH_{jkt}, due to the backwater effects caused by a dam located at downstream site k.

In this model, where the head height is not constant and hence the hydropower potential of the sites is not fixed, parameter w_{ji} can no longer be used to express the actual hydropower potential of site j. Instead, the hydropower potential of each site should be defined dynamically, capturing the potential changes in head height that installation decisions hold. For this reason a new decision variable, π_{ji}, is introduced to reflect the actual hydropower potential of site j.

In particular, given the following additional decision variables:

$\pi_{ji} =$ hydropower potential at site j given installation of dam size i

$$\lambda_{jkt} = \begin{cases} 1 & \text{if a dam of size } t \text{ installed at site } k \text{ has a backwater effect on a dam} \\ & \text{located at site } j \text{ upstream} \\ 0 & \text{otherwise} \end{cases}$$

a more general formulation of HPLP is given by:

$$[\text{HPLP2}]\ \max \sum_{j \in N} \sum_{i \in S_j} \pi_{ji} \tag{2.13}$$

subject to constraints (2.4) - (2.12) and the following:

$$\pi_{ji} \leq a_{ji} H_{ji} x_{ji} \qquad \forall j \in N, i \in S_j \tag{2.14}$$

$$\pi_{ji} \leq a_{ji} H_{ji} - a_{ji} \sum_{k \in M_j} \sum_{t \in S_k} \Delta H_{jkt} \lambda_{jkt} \qquad \forall j \in N, i \in S_j \tag{2.15}$$

$$\lambda_{jkt} \geq \sum_{i \in S_j} x_{ji} + x_{kt} - 1 - \sum_{\ell \in I_{jk}} \sum_{s \in S_\ell} x_{\ell s} \qquad \forall j \in N, k \in M_j, t \in S_k \tag{2.16}$$

$$\lambda_{jkt} \geq 0 \qquad \forall j \in N, k \in M_j, t \in S_k \tag{2.17}$$

$$x_{ji} + x_{kt} \leq 1 + \sum_{\ell \in I_{jk}} \sum_{s \in S_\ell} x_{\ell r} \qquad \forall j \in N, i \in S_j, k \in M_j, t \in S_k | \Delta H_{jkt} \geq H_{ji} \tag{2.18}$$

The objective function (2.13), similar to (2.2), maximizes total hydropower potential. The difference from (2.2) is that hydropower is no longer fixed for each location and dam size option, hence the use of decision variables π_{ji}. Inequalities (2.14) and (2.15), in combination, determine the hydropower potential of each site j, where parameter $a_{ji} = \eta_{ji} \rho g Q_j$. Specifically, if no dam is located at site j, constraints (2.14) forces hydropower potential to be 0. On the other hand, if a dam of size i is located at site j, (2.15) becomes strictly binding and specifies that the hydropower potential of the dam must be less than or equal to the power that can be produced with a nominal head value of H_{ji} minus any decrease in power caused by the existence of a backwater effect on site j (i.e., if $\lambda_{jkt} = 1$, for any $k \in M_j, t \in S_k$, a head reduction of ΔH_{jkt} occurs). Note that if $x_{ji} = 1$ and there is no backwater effect on site j, then (2.14) and (2.15) will be binding. Constraints (2.16) guarantee that $\lambda_{jkt} = 1$ if and only if a hydropower dam is installed at j, a dam of size t is installed at k, and no dam is installed in between them (i.e., $\sum_{\ell \in I_{jk}} \sum_{s \in S_\ell} x_{\ell s} = 0$). For all other situations, constraints (2.17) prevent λ_{jkt} from becoming negative. Due to the structure of the problem, the λ_{jkt} variables are guaranteed to take on binary values. The next set of constraints (2.18) prevent the nonsensical siting of dams in which the installation of a dam would cause an upstream dam to become completely "swamped" (i.e., the reduction in head ΔH_{jkt} caused by a backwater effect is greater than the initial head H_{ji} of the dam).

2.2.4 Backwater Effects on Barrier Passability

In the above model [HPLP2], it is inherently assumed that backwater effects only impact hydropower potential. In the majority of cases, particularly for small dams and weirs, head is also a critical factor in determining the passability of a barrier. In what follows, we present an even more general model, denoted [HPLP3], in which backwater effects can also influence the passability of barriers. To begin with, assume that passability p_j at site j is determined by the function:

$$p_j = f_j(H_j, \mathbf{x}_j) \qquad \forall j \in J \tag{2.19}$$

where H_j is the effective head height at site j and $\mathbf{x}_j = \left(x_{j1}, \ldots x_{j|S_j|}\right)$ specifies the vector of hydropower dam installation decisions for site j. Note that the p_j variables would, in turn, influence cumulative passabilities such that $z_j = \prod_{k \in D_j} p_k$, $\forall j$. In the special case where equations (2.19) form a set of step-functions (e.g., equation (2.34) used in our case-study described below), it is possible to formulate a linear model using a piece-wise linear representation of (2.19), as described in Winston (2004), Sec. 9.2.

Specifically, let H_j^0 be the initial head height for site j and let H'_{ji} be the nominal increase in head height due to the installation of a dam of size i at site j. As before, ΔH_{jkt} represents the change in head height due to the backwater effect caused by a dam of size t located at site k downstream. Further, let $\hat{p}_{jr}^0$, $r = 1, \ldots, R$, be the nominal passability level of site j when no hydropower dam is located at j and head height H_j is in the range $\left(\hat{H}_r, \hat{H}_{r+1}\right]$. Similarly, let $\hat{p}_{jir}$ be the passability of site j when a dam of size i is built there and head height H_j is in the range $\left(\hat{H}_r, \hat{H}_{r+1}\right]$. Note that the $\hat{H}_r$ define a total of $R+1$ breakpoints along the curve H_j versus $f_j(H_j, \mathbf{x}_j)$. By introducing the following auxiliary variables:

$$\mu_{jkt} = \begin{cases} 1 & \text{if a dam of size } t \text{ is installed at } k \text{ and no dam is installed between } j \text{ and } k \\ 0 & \text{otherwise} \end{cases}$$

$$u_{jr} = \begin{cases} 1 & \text{if the head height for site } j \text{ is in the range } \left(\hat{H}_r, \hat{H}_{r+1}\right] \\ 0 & \text{otherwise} \end{cases}$$

$$\theta_{jr} = \text{weight assigned to } r\text{-th breakpoint } \hat{H}_r \text{ for site } j$$

$$\psi_{jr} = \text{cumulative passability of site } j \text{ given that } j\text{'s head height is in the range } \left(\hat{H}_r, \hat{H}_{r+1}\right]$$

equations (2.19) can be replaced with (2.20) - (2.27) below.

Determination of Head Height

$$\sum_{r=1}^{R+1} \hat{H}_r \theta_{jr} = H_j^0 + \sum_{i \in S_j} H'_{ji} x_{ji} - \sum_{k \in M_j} \sum_{t \in S_k} \Delta H_{jkt} \mu_{jkt} \qquad \forall j \in J \tag{2.20}$$

$$\sum_{r=1}^{R+1} \theta_{jr} = 1 \qquad \forall j \in J \tag{2.21}$$

$$\theta_{jr} \geq 0 \qquad \forall j \in J, r = 1, \ldots, R \tag{2.22}$$

$$\theta_{jr} \leq \begin{cases} u_{jr} & r = 1 \\ u_{j(r-1)} + u_{jr} & r = 2, \ldots, R-1 \\ u_{j(r-1)} & r = R+1 \end{cases} \qquad \forall j \in J \tag{2.23}$$

$$\sum_{r=1}^{R} u_{jr} = 1 \qquad \forall j \in J \tag{2.24}$$

$$u_{jr} \in \{0, 1\} \qquad \forall j \in J, r = 1, \ldots, R \tag{2.25}$$

$$\mu_{jkt} \leq x_{kt} \qquad \forall j \in J, k \in M_j, t \in S_k \tag{2.26}$$

$$\mu_{jkt} \leq 1 - \sum_{s \in S_\ell} x_{\ell s} \qquad \forall j \in J, k \in M_j, t \in S_k, \ell \in I_{jk} \tag{2.27}$$

Equations (2.20) in combination with constraints (2.21) and (2.22) simply require that a convex combination of the breakpoints $\hat{H}_r$ with weights θ_{jr} (the left hand side of (2.20)) be found which is equal to the effective head height of site j (the right hand side (2.20)). The effective head height at site j, in turn, is equal to the initial head H_j^0 plus any nominal increase in head H'_{ji} due to the installation of a dam of size i ($x_{ji} = 1$) minus any decrease in head ΔH_{jkt} due to the backwater effect on site j caused by a dam of size t located at downstream site k ($\mu_{jkt} = 1$). Constraints (2.23) - (2.25) enforce adjacency restrictions on the θ_{jr} weighting variables, namely that at most two weights can be positive and must be adjacent. Assuming that passability and head height are inversely related, it is preferable, all things considered, for $\mu_{jkt} = 1$ in order to have higher passability at site j and so more easily meet the minimum accessible habitat requirement (2.4). Constraints (2.26) and (2.27) force variable μ_{jkt} to be equal to 0 if either no dam is located at site k downstream ($x_{tk} = 0$) or a dam is installed between k and j ($\sum_{s \in S_\ell} x_{\ell s} = 1 | \ell \in I_{jk}$) .

Given a correct determination of the head height at site j, the u_{jr} can be used to determine the cumulative passability of site j through the use of constraints (2.28) - (2.32) below.

Determination of Passability

$$z_j = \sum_{r=1}^{R} \psi_{jr} \qquad \forall j \in J \tag{2.28}$$

$$\psi_{jr} \leq \hat{p}^0_{jr} u_{jr} + \sum_{i \in S_j} x_{ji} \qquad \forall j \in J, r = 1, \ldots, R \tag{2.29}$$

$$\psi_{jr} \leq \hat{p}^0_{jr} z_{d_j} + \sum_{i \in S_j} x_{ji} \qquad \forall j \in J | D_j \neq \emptyset, r = 1, \ldots, R \tag{2.30}$$

$$\psi_{jr} \leq \hat{p}_{jir} u_{jr} + 1 - x_{ji} \qquad \forall j \in N, i \in S_j, r = 1, \ldots, R \tag{2.31}$$

$$\psi_{jr} \leq \hat{p}_{jir} z_{d_j} + 1 - x_{ji} \qquad \forall j \in N | D_j \neq \emptyset, i \in S_j, r = 1, \ldots, R \tag{2.32}$$

Equations (2.28) determine the cumulative passability of each site j by summing across the cumulative passability terms ψ_{jr} associated with head height ranges r. For each head height interval $\left(\hat{H}_r, \hat{H}_{r+1}\right]$, $r = 1, \ldots, R$, inequalities (2.29) and (2.30) set upper bounds on the cumulative passability value ψ_{jr} when no dam is located at site j ($\sum_{i \in S_j} x_{ji} = 0$), while inequalities (2.31) and (2.32) apply if a dam of size i is located at j ($x_{ji} = 1$). Given that exactly one of the u_{jr} variables will be equal to 1 (i.e., head height must fall within a specific range $\left(\hat{H}_r, \hat{H}_{r+1}\right]$), a single pair of constraints, either (2.29) - (2.30) or (2.31) - (2.32) depending on the dam installation decision, will be binding for each site j. Regardless of the dam location decision for site j, constraints (2.29) - (2.30) and (2.31) - (2.32) work in the exact same fashion as (2.11) - (2.12) do for the simpler model [HPLP2], in which backwater effects on passability are ignored. More specifically, they form a series of probability chains that iteratively evaluate cumulative barrier passability by starting from the most downstream barrier and progressively moving to barriers upstream.

We note that the above linearization is actually quite general. Even when equation (2.19) is not strictly a step-wise function, it is possible to approximate a continuous nonlinear curve to any degree of accuracy by introducing a sufficient number of breakpoints R and auxiliary u_{jr}, θ_{jr}, and ψ_{jt} variables and constraints.

2.3 Case Study

2.3.1 Background

A case study of England and Wales will be used to illustrate the benefits of using our proposed framework. We started with a dataset consisting of the location of 25,935 natural (i.e., waterfalls) and artificial (i.e., weirs, dams, barrages, and locks) barriers compiled by the UK Environment Agency (EA, 2010). Each barrier in

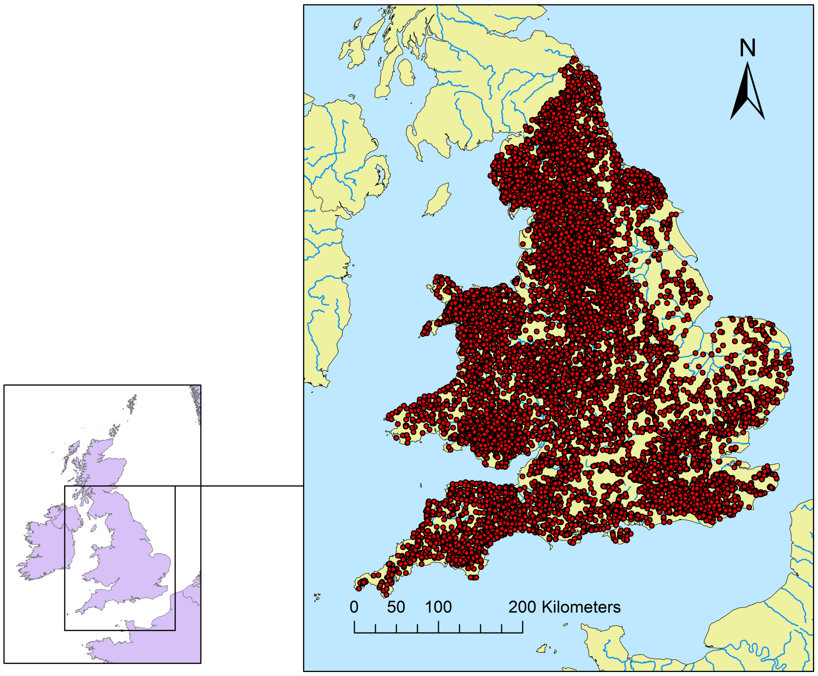


Figure 2.2: Location of artificial and natural fish barriers across England and Wales.

the EA's database is georeferenced and includes a description of its barrier type and head value. These head values correspond to the differences between the upstream and downstream water elevations of the barriers and were obtained from aerial surveys using a combination of Light Detection and Ranging (LIDAR) and Interferometric Synthetic Aperture Radar (IFSAR) remote sensing technology (EA, 2010).

In order to extract all the necessary input values for the optimization models a series of data processing steps had to take place. First, in order to determine key barrier parameters, including each barrier's immediate downstream barrier (d_j) and net upstream river length (v_j), we used the RivEX toolbox (Hornby, 2014) for ArcGIS 10.2.1. Working off a 1:50,000 scale continuous center-line hydrology layer provided by the UK Centre for Ecology and Hydrology (CEH) (Moore et al., 1994), we first generated a single-threaded river network. The barrier points were subsequently snapped to the river network using a 50m snapping distance. This resulted in a final dataset of 14,682 artificial and 4,947 natural barriers, as shown in Figure 2.2.

Following common practice within England and Wales, we assumed that SHP could only be installed at existing dam/weir sites. We considered three different SHP sizing options. All dams/weirs with head heights up to 5m were deemed suitable for the installation of a 5m SHP; those with heights between 5 and 10m were candidates for a 10m SHP. For any dam/weir with a height greater than 10m, installation of an SHP was assumed to not increase the existing height of the structure. As a conservative estimate (Cyr et al., 2011), we assumed SHPs had a conversion efficiency of $\eta_{ji} = 0.7$, $\forall j, i$.

In the next data processing step, in order to determine flow values (Q_j) of each SHP candidate site, we developed a regression model to predict mean flow based on mean annual precipitation within the site's upstream catchment area. Mean flow data were obtained for 1,403 georeferenced gauging stations from the UK National River Flow Archive (NRFA). In a series of ArcGIS steps, we delineated the catchment area for each gauging station using 50m digital elevation model (DEM), outflow drainage direction, and cumulative catchment area grids provided by CEH. Gauging station catchment areas were then overlaid on a 5km×5km annual precipitation grid for England and Wales produced from UK MetOffice historical monthly average rainfall grids for the period 1981-2010 (MetOffice, 2014). From this, area-weighted annual precipitation could be determined for each gauging station ($precip_j$) and subsequently used to estimate mean flow (Q_j) as follows.

$$\ln(Q_j) = -8.37 + 1.05 \ln(precip_j) \tag{2.33}$$

The log-linear model (2.33) produced a very good fit to the data, with an adjusted R^2 of 0.89. The previous GIS steps were then repeated to calculate a $precip_j$ value for each potential SHP site j and estimate an associated flow volume Q_j based on regression model (2.33).

Another essential data processing step was to determine the potential changes in head height due to the backwater effect of an SHP located downstream (ΔH_{jkt}). This required a series of substeps described below. Under a gradually varied flow regime, backwater profiles for each SHP site up to the nearest SHP or river confluence point can be found using the "standard step" method, as described in Chadwick et al. (2013). This method allows the evaluation of depth at any specified distance upstream of a structure by dividing the watercourse into equal intervals and then iteratively calculating depth at upstream cross sections by solving an energy balance governing equation (Chow, 1959). The standard step method requires, among other things, information about the slope and channel geometry of each upstream cross section. Slope values were calculated in ArcGIS using the DEM provided by CEH. We assumed that watercourses had a simple rectangular geometry. Stream width was estimated based on a river segment's Strahler stream order. To do this, we determined using RivEX the Strahler order (a gross measure of stream size) for each stream segment in the CEH river network and then overlaid the locations of 24,130 field measurements of stream width taken across the UK (M. Naura, University of Southampton, pers. comm.) to produce a look-up table of Strahler order versus mean stream width.

Finally, we assumed that SHPs would be fitted with fish passes having a combined upstream/downstream passage efficiency of 0.5. This is broadly in line with the findings of Noonan et al. (2012). For a site where no SHP is installed, passability was assumed to vary with the height of a barrier. Based on protocols developed

Table 2.1: Model sizes of [HPLP1] and [HPLP3].

Model	Variables		Constraints
	Binary	Continuous	
[HPLP1]	$\sum_{j\in N}\lvert S_j\rvert$	$\lvert J\rvert+\sum_{j\in N}\lvert S_j\rvert$	$1+\lvert N\rvert+\lvert J\rvert+\sum_{j\in N^0}\lvert S_j\rvert+\sum_{j\in N^{1^+}}\lvert S_j\rvert$
[HPLP3]	$\sum_{j\in N}\lvert S_j\rvert+R\lvert J\rvert$	$2(R+1)\lvert J\rvert+\sum_{j\in N}\lvert S_j\rvert$ $+2\sum_{j\in N}\sum_{k\in M_j}\lvert S_k\rvert$	$1+3\lvert N\rvert+(2R+5)\lvert J\rvert+2\sum_{j\in N}\sum_{k\in M_j}\lvert S_k\rvert$ $+R\sum_{j\in N^{1^+}}\lvert S_j\rvert+\sum_{j\in J}\sum_{k\in M_j}\lvert S_k\rvert\lvert I_{jk}\rvert+\lvert A\rvert$

Where set N^0denotes candidate dam sites with no immediate downstream barrier (i.e., $N^0=\{j\in N|D_j=\emptyset\}$), sets $N^{1^+}$and J^{1^+} denote, respectively, candidate dam sites and all potential barrier sites with at least one downstream barrier (i.e., $N^{1^+}=\{j\in N|D_j\neq\emptyset\}$ and $J^{1^+}=\{j\in N|D_j\neq\emptyset\}$) and set A is defined as $A=\{j\in N,k\in M_j,i\in S_j,t\in S_k|\Delta H_{jkt}\geq H_{ji}\}$.

Table 2.2: Model sizes of [HPLP1] and [HPLP3] for the England and Wales case study area.

Model	Variables		Constraints
	Binary	Continuous	
[HPLP1]	14,682	34,311	62,495
[HPLP3]	93,198	299,064	524,124

in SNIFFER (2010) for adult trout, we used the following to determine upstream passability p as function of head height H.

$$p=\begin{cases}1 & \text{if } H\leq 0.4\text{ m}\\ 0.6 & \text{if } 0.4\text{ m}<H\leq 0.6\text{ m}\\ 0.3 & \text{if } 0.6\text{ m}<H\leq 1\text{ m}\\ 0 & \text{if } H>1\text{ m}\end{cases}\tag{2.34}$$

Based on this, we used a set of $R+1=5$ breakpoints to define equation (2.20), such that $\hat{\mathbf{H}}=\{-6,0.4,0.6,1,75\}$ and $\hat{\mathbf{p}}^0=\{1,0.6,0.3,0\}$. The first breakpoint (-6m) corresponds to the largest (negative) change in head value due to swamping, while the last breakpoint (75m) corresponds to the largest head height observed in our dataset.

2.3.2 Results

The basic model [HPLP1] and the backwater effects model [HPLP3] were both implemented in C++ using CPLEX callable libraries version 12.6. All experiments were performed on the same quad-core Dell OptiPlex 9020 laptop (Intel i7-4770 processor, 3.4 GHz per chip) with 8GB of RAM and running Windows 7 64-bit operating system. Model sizes of [HPLP1] and [HPLP3] can be found in Table 2.1 while the specific sizes for our case study area are reported in Table 2.2.

Before going into our analysis, it is important to point out that river connectivity within England and Wales is heavily impaired by the presence of existing barriers. Only about 3% (3,410 km) of the 132,071 km of potential stream habitat located above barriers is currently accessible to migratory fish. In systems with few existing barriers, minimum accessible habitat requirements would normally ensure that comparatively small numbers of dam are installed. In our case study, however, there are nearly 20,000 existing barriers, the majority of which (75%) are completely impassable. According to model [HPLP3] with $\alpha = 0$, up to 14,607 SHPs could be installed across England and Wales, resulting in a maximum hydropower potential of 691.9 MW, while at the same time increase accessible habitat by 229% to 11,217 km of river.

To consider a more realistic scenario of hydropower development, we added the following constraint to both [HPLP1] and [HPLP3]:

$$\sum_{j \in N} \sum_{i \in S_j} x_{ji} \leq n \tag{2.35}$$

which allowed us to determine what the maximum hydropower production would be if at most n new SHPs were located. In addition, we observed during preliminary experiments that both [HPLP1] and [HPLP3] occasionally selected sites with unrealistically small hydropower potential (i.e.,$\ll$1 kW), mainly in an attempt to satisfy the minimum accessible habitat requirement (2.4). Indeed, a quick inspection of the England and Wales dataset reveals that among the 14,682 candidate dam sites, nearly a quarter (3,557) have hydropower potential less than 1 kW. In practice, development of sites with insufficient hydropower potential is difficult to justify on economic grounds. To prevent the selection of low-hydropower sites, therefore, we added the following set of constraints to [HPLP1]:

$$cx_{ji} \leq w_{ji} \qquad \forall j \in N, i \in S_i \tag{2.36}$$

and an equivalent set of constraints to [HPLP3]:

$$cx_{ji} \leq \pi_{ji} \qquad \forall j \in N, i \in S_i \tag{2.37}$$

In our implementation, we set constant $c = 5000$, thus excluding all sites with hydropower potential <5 kW (typically termed "pico" hydro scale plants). Adding minimum site-level hydropower constraints (2.37) to [HPLP3] with constraint (2.35) non-binding (e.g., $n = 14,628$) and $\alpha = 0$, a total of 7,672 SHPs could be installed, resulting in a maximum hydropower potential of 681.9 MW and a 177% increase in accessible habitat (9,439 km total).

Table 2.3: Hydropower potential and accessible habitat for various SHP development scenarios.

	[HPLP1]			[HPLP3]		
n	Hydropower (MW)	Habitat (km)	Time (s)	Hydropower (MW)	Habitat (km)	Time (s)
≥0% Accessible Habitat Increase ($\alpha = 1.0$)						
100	174.7	3,935	4.7	174.4	4,027	234.2
500	368.5	4,532	4.6	365.4	4,592	161.1
1,000	471.1	5,302	4.5	465.4	5,345	199.5
≥50% Accessible Habitat Increase ($\alpha = 1.5$)						
100	173.2	5,119	5.9	172.9	5,119	531.7
500	367.9	5,145	5.5	364.9	5,116	728.9
1,000	471.1	5,302	4.6	465.4	5,345	216.1
≥100% Accessible Habitat Increase ($\alpha = 2.0$)						
100	155.3	6,821	8.9	154.7	6,821	936.2
500	362.1	6,828	8.7	359.1	6,822	653.1
1,000	469.4	6,827	7.6	463.7	6,821	698.4
≥150% Accessible Habitat Increase ($\alpha = 2.5$)						
100	-	-	-	-	-	-
500	342.8	8,526	15.1	339.0	8,526	1518.0
1,000	461.0	8,530	10.8	451.5	8,527	889.9
≥200% Accessible Habitat Increase ($\alpha = 3.0$)						
100	-	-	-	-	-	-
500	284.5	10,231	25.3	283.9	10,231	2412.5
1,000	437.6	10,232	12.6	429.9	10,231	2250.1

A '-' indicates that no feasible solution could be obtained for a given model due to the minimum accessible habitat requirement (2.4).

Table 2.3 reports hydropower potential, accessible habitat, and run times (in CPU seconds) for models [HPLP1] and [HPLP3] given the installation of 100, 500, or 1,000 new SHPs. It is interesting to note that a small subset of candidate sites accounts for a large portion of hydropower generation potential within the study area. For example, according to [HPLP3], almost 25% of maximum hydropower generation capacity (174.4 MW) can be achieved by siting 100 SHPs, which corresponds to just 0.6% of all candidate sites. With 1,000 dams (6.8% of all candidate sites), almost 67% of maximum hydropower development potential (465.4 MW) can be achieved.

What really stands out from analyzing Table 2.3 is that for our particular study area the installation of hydropower dams actually creates a "win-win" situation with regards to increasing renewable energy production and improving river connectivity. Assuming that an SHP is equipped with even a moderately efficient fish pass (0.5 passability), the requirement for a ≥100% increase in accessible habitat (equivalent to more than 6,800 river km) could be met according to either model [HPLP1] or [HPLP3]. With 500 or 1,000 SHPs, requirements for either a ≥150% or ≥200% increase in fish habitat would be satisfied. Nonetheless, there are distinct tradeoffs between increasing fish habitat, on the one hand, and achieving maximum hydropower potential, on the other. Figure 2.3 shows how hydropower potential deceases with increases in accessible

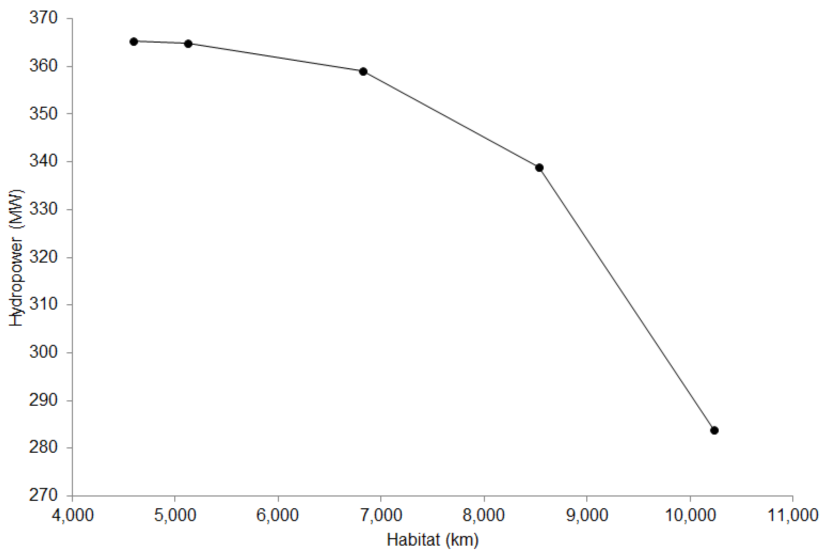


Figure 2.3: Maximum hydropower potential considering backwater effects (model [HPLP3]) versus total accessible habitat given 500 SHPs.

habitat given the location of 500 SHPs.

A comparison of [HPLP1] and [HPLP3] shows that ignoring backwater effects results in a small to moderate overestimation of maximum hydropower potential regardless of accessible habitat requirements. This overestimation goes up as the number of SHPs increases. For example, when no increase in accessible habitat is required, the difference in hydropower potential for [HPLP3] given 100 SHPs is a mere -0.3 MW (-0.2%). When the number of barriers increases to 1,000, however, there is a -5.7 MW (-1.2%) difference in hydropower for [HPLP3]. The largest difference (-9.5 MW) is observed for 1,000 dams and a $\geq$150% increase in accessible habitat requirement. The observation that [HPLP1] always suggests solutions with higher hydropower output can be explained by the fact that [HPLP3] has a lot more constraints.

What is also clear from looking at Table 2.3 is that including backwater effects can result in an appreciable increase in solution time. Regardless of the number of dams or accessible habitat requirements, [HPLP1] can be solved in a matter of seconds to 10s of seconds. For [HPLP3], times vary from several minutes (100 SHPs and a $\geq$0% increase in accessible habitat) to over 40 minutes (500 SHPs and a $\geq$200% increase in accessible habitat). This difference in solution time is expected given the difference in model sizes presented in Tables 2.1 and 2.2.

For both models the 100 SHP scenario is infeasible for an 150% increase in habitat and over. This infeasibility is not surprising as 100 hydropower dams are relatively few to achieve so high level of accessible habitat increase. More hydropower dams would be needed to satisfy requirements of this level, which is the case with the 500 and 1000 SHPs with which feasible solutions are reached.

Table 2.4: Variation in hydropower potential (in MW) for models [HPLP1] and [HPLP3] with and without backwater effects included.

	Solutions to [HPLP1]		Solutions to [HPLP3]	
n	Without Backwater$^+$	With Backwater	Without Backwater	With Backwater$^+$
		≥0% Accessible Habitat Increase ($\alpha = 1.0$)		
100	174.7	173.6	174.4	174.4
500	368.5	Infeas.	365.4	365.4
1000	471.1	Infeas.	467.1	465.4
		≥50% Accessible Habitat Increase ($\alpha = 1.5$)		
100	173.2	172.1	172.9	172.9
500	367.9	Infeas.	364.9*	364.9
1000	471.1	Infeas.	467.1	465.4
		≥100% Accessible Habitat Increase ($\alpha = 2.0$)		
100	155.3	154.2	154.7	154.7
500	362.1	Infeas.	359.1*	359.1
1000	469.4	Infeas.	465.5*	463.7
		≥150% Accessible Habitat Increase ($\alpha = 2.5$)		
100	-	-	-	-
500	342.8	Infeas.	339.0*	339.0
1000	461.0	Infeas.	453.0*	451.5
		≥200% Accessible Habitat Increase ($\alpha = 3.0$)		
100	-	-	-	-
500	284.5	Infeas.	283.9*	283.9
1000	437.6	Infeas.	431.0*	429.9

A'+' indicates the original solution. A '-' indicates that no feasible solution to the original model could be obtained due to the minimum accessible habitat requirement (2.4). For solutions to [HPLP1], 'Infeas.' indicates that one or more swamping constraints (2.18) are violated when backwater effects are included. For solutions to [HPLP3], a '*' indicates that the minimum accessible habitat requirement (2.4) is not strictly satisfied when backwater effects are ignored.

Table 2.4 shows how hydropower potential varies for models [HPLP1] and [HPLP3] with and without backwater effects included (i.e., by plugging solutions from [HPLP1] into [HPLP3] and vice versa). It is interesting to note that in spite of the relatively modest backwater effects predicted for our case study area, the vast majority of solutions to [HPLP1] (10 out of 13) are infeasible with respect to the non-swamping constraints (2.18), meaning one or more dams would end up being submerged due to the presence of a downstream dam. It is also interesting that more than half of [HPLP3] solutions (7 out of 13) would be technically infeasible, due to violations of the minimum accessible habitat requirement (2.4), if backwater effects were ignored. This occurs because small but material increases in accessible habitat (0.1-0.9%) are produced when passability is calculated dynamically as function of head height (via constraints (2.20) - (2.32)), thus allowing accessible habitat requirements to just be met by solutions to [HPLP3].

Table 2.5 reports basic statistics about initial head height, Strahler stream order, and distance to river mouth of SHP sites selected by [HPLP3] for various minimum accessible habitat requirements. Column "All" refers to all 14,682 candidate sites. What stands out is that low-head dam/weir sites (≤5 m) are far and away the preferred choice for siting SHPs. Such sites make up roughly 87% of all artificial barriers, but account for no less than 95% of selected sites, regardless of the specific number of SHPs sited or minimum requirements on accessible habitat.

Another observation is that selected SHPs tend to be located on high-order streams. This is not at all surprising given that stream order is normally a very good proxy for flow (Q) and, in turn, hydropower potential (π). Looking at the various solutions in Table 2.5, SHPs are never located on order 1-2 streams nor even on order 3 streams unless 1,000 SHPs are located. Instead, the vast majority (89-100%) of SHPs are located on order 5-7 streams.

What is more interesting is that for any given number of SHP sites, model [HPLP3] selects locations that are both closer to the river mouth and on lower order streams as the minimum accessible habitat requirement increases. Given 100 SHPs, for example, average distance to mouth decreases by 16.8 km (from 105.8 km to 89.0 km) when the accessible habitat requirement changes from $\geq$0% to $\geq$100%. At the same time, the number of sites selected on mid order 3-4 streams goes from 0 to 10.

Locating SHPs fitted with fish passes closer to the river mouth makes perfect sense if the primary aim is to increase accessible river habitat; barriers closer to the sea will generally disrupt longitudinal river connectivity the most. However, within a given river catchment, stream order and distance to mouth are normally inversely related, with low order streams found higher up in the catchment (i.e., further away from the river mouth). All thing being equal then, the *a priori* hypothesis would be that sites on mainstem, high-order rivers that are also close to the sea should be preferred.

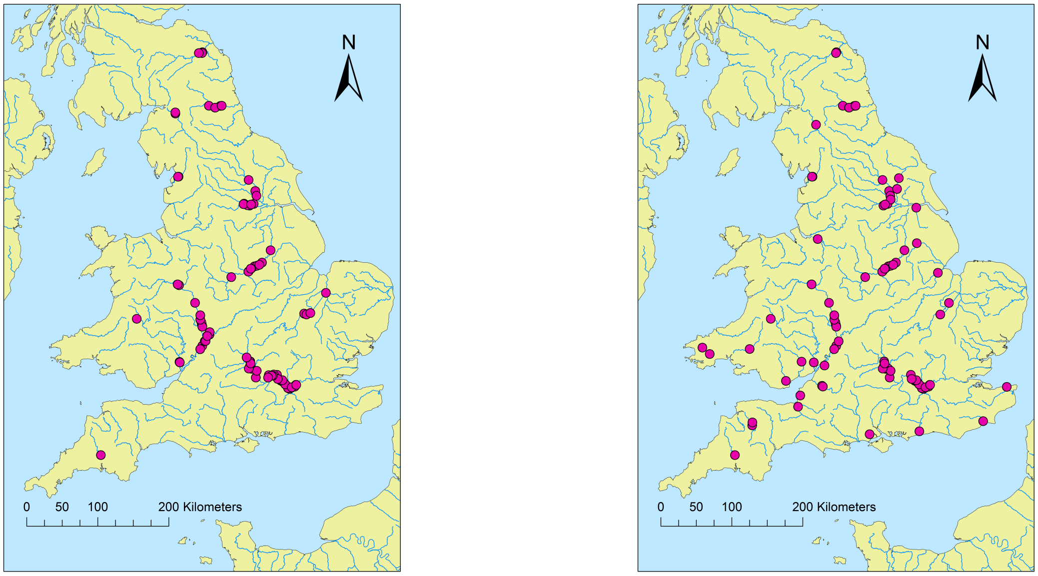
This apparent contradiction is explained by the shifting spatial pattern of SHP location. Inspection of Figure 2.4 shows that SHPs are predominately located on major, high-order rivers, such as the Thames, the Severn, the Trent, the Aire, the Tyne, and their major tributaries when habitat requirements are less stringent (i.e., given a 0% or 100% minimum increase in accessible habitat). However, when habitat requirements are at the high end (i.e., given a 100% minimum increase in accessible habitat), many more SHPs are located on smaller, middle-order rivers at sites closer to the sea. Ultimately, what this shows is that balancing tradeoffs between hydropower and river connectivity is a complex issue. Depending on one's aims, the best locations for hydropower development can vary considerably.

2.4 Conclusions

Proposals to install hydropower dams inevitably raise conflict between the need for renewable energy production on the one hand and the desire for maintaining healthy, well-connected river ecosystems on the other. In this chapter, we present a suite of optimization based tools for locating small hydropower dams in an

Table 2.5: Key attributes of selected SHP sites given n =100, 500, or 1,000 and minimum accessible habitat increases of ≥0, 50, or 100%.

		Accessible Habitat Increase								
		≥0%			≥50%			≥100%		
	All	$n = 100$	$n = 500$	$n = 1,000$	$n = 100$	$n = 500$	$n = 1,000$	$n = 100$	$n = 500$	$n = 1,000$
Initial Head Height (H)										
H≤5 m	12,741	95	489	959	95	490	959	95	491	960
5 m < H≤10 m	1,562	4	6	30	4	6	30	4	5	30
H > 10 m	379	1	5	11	1	4	11	1	4	10
Strahler Order										
1	2,811	-	-	-	-	-	-	-	-	-
2	3,696	-	-	-	-	-	-	-	-	-
3	4,114	-	-	6	-	-	6	-	-	6
4	2,381	-	12	102	-	13	102	10	18	106
5	1,143	1	100	409	3	102	409	9	107	408
6	453	45	306	401	43	303	401	30	294	398
7	84	54	82	82	54	82	82	51	81	82
Avg Dist to Mouth (km)	96.3	105.8	111.9	108.7	104.1	111.0	108.7	89.0	109.8	107.9



(a) (b)

Figure 2.4: Solutions to the backwater effects model [HPLP3] with 100 SHPs given a ≥0% (a) and ≥100% (b) increase in accessible habitat.

environmentally friendly manner. Importantly, we take into account the backwater effects that dams have on both hydropower and fish passability at nearby upstream sites. Through a series of linearization steps, we manage to formulate a mixed-integer linear programing model.

The usefulness of our framework is demonstrated with a case study from England and Wales. We find that our backwater effects model is highly scalable. With more than 14,000 candidate sites, model [HPLP3] could still solve in less than an hour, regardless of accessible habitat requirements. One key result is that a comparatively small number of sites accounts for a large portion of hydropower potential within the study area. Installation of just 100 SHPs can produce 25% of maximum hydropower generation capacity, while 67% of maximum hydropower can be achieved by siting 1,000 SHPs. More importantly, given the heavily impaired state of river connectivity across England and Wales, installation of SHP can actually create a win-win result yielding both increased hydropower and improved river connectivity if SHPs are fitted with even moderately effective fish passes. We also observe that optimal SHP locations vary depending on how stringent requirements are for increasing amounts of accessible river habitat. SHPs are predominately located in large river systems when habitat requirements are low to moderate and more frequently in smaller river systems when habitat requirements are high.

In our case study, we found that backwater effects had only a modest influence on maximum hydropower potential and accessible river habitat. It is important to emphasize, however, that the extent of backwater effects will be context dependent, determined in large part based on the size and spacing of dams and the geometry of river channels. Across England and Wales, river connectivity and water surface profiles are already heavily impacted by a large number of existing barriers. Moreover, we assumed that 1) hydropower facilities could only be installed at existing weirs and 2) increases in head height were restricted to ≤ 5 m. Consequently, even though spacing among candidate SHP sites is tight along some stretches of river, backwater effects were not as pronounced compared to a situation where dams could be constructed at "greenfield" sites (i.e., where barriers are not currently present). In addition, many river channels across England and Wales have relatively steep slopes (critical depth > normal depth), which causes a backwater curve to reduce in length. Indeed, for most SHP candidate sites in our study, the backwater curve did not extend to any immediate upstream sites due to the steepness of the channel slope. In other study areas, where such conditions do not hold, we would expect backwater effects to have a much larger impact on hydropower potential and accessible habitat.

Regardless of the relative influence of backwater effects on hydropower and river connectivity, our results clearly show the benefit of taking backwater effects into account. Solutions to our simpler model [HPLP1], which ignored backwater effects, frequently produced infeasible solutions in which a dam would be entirely swamped due to the presence of a nearby dam downstream. Hence, even though our more complex model [HPLP3] had a marked overhead in terms of solution times, it invariably produced more realistic solutions that did not violate non-swamping constraints.

We acknowledge that a more in-depth case study would include cost information related to the construction of dams and fish passes, as well as the monetary benefits of hydropower production. Unfortunately, this goes beyond the scope of our present study. While fish pass costs can be estimated fairly accurately based on the height of a structure, dam construction costs vary considerably from site to site depending on the structural characteristics of any existing weir and the geology/topology of the surrounding area. Devising realistic cost estimates is thus difficult without conducting extensive field surveys. Moreover, we believe our model is primarily suited to the strategic level needs of environmental/energy planning authorities concerned with where hydropower development should be permitted while limiting impacts on river connectivity. Given this, the main focus of our case study is on analyzing hydropower potential across England and Wales rather than performing a detailed economic analysis of the costs and benefits that would accrue to individual companies (usually privately owned) who would ultimately build and operate hydropower facilities.

There are a number of ways in which our models could be extended. For example, we could adapt our modeling framework to handle potadromous dispersal patterns, to focus on larger, reservoir type dams or to consider hydropower dam placement together with artificial barrier mitigation decisions. A detailed discussion regarding potential future research can be found in Chapter 5.

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Chapter 3

The Importance of Spatiotemporal Fish Population Dynamics in Barrier Mitigation Planning

In this study, we propose a novel optimization framework to prioritize fish passage barrier mitigation decisions that incorporates both fish population and dispersal dynamics in order to maximize equilibrium population size. A case study involving a wild coho salmon (*Oncorhynchus kisutch*) population from the Tillamook basin, Oregon, USA is used to illustrate the benefits of our approach. We consider two extreme homing patterns, river and reach level homing, as well as straying. Under density dependent population growth, we find that the type of homing behavior has a significant effect on barrier mitigation decisions. In particular, with reach homing, our model results in virtually the same population sizes as a more traditional barrier prioritization procedure that seeks to maximize the accessible habitat. With river homing, however, there is no need to remove all barriers to maximize equilibrium population size. Indeed, a stochastic version of our model reveals that removing all barriers actually results in a marginal increase in quasi-extinction risk. We hypothesize that this is due to a population thinning effect of barriers, resulting in a surplus of recruits in areas of low spawner density. Our present study should prove useful to fish conservation managers by assessing the relative importance of incorporating spatiotemporal fish population dynamics in river connectivity restoration planning.

3.1 Introduction

Artificial in-stream barriers, such as culverts, dams, and weirs, can hinder or altogether prevent migratory fish from reaching essential breading and rearing habitats (O'Hanley and Tomberlin, 2005; King and O'Hanley, 2016), resulting in restricted range, reduced productivity, and cascading changes in aquatic community composition (Stanford et al., 1996). Reconnecting stream habitats isolated by the presence of so called fish passage barriers is widely considered a top priority for restoring healthy fish populations (Roni et al., 2008; Kemp and O'Hanley, 2010).

In this study, we propose a framework for prioritizing barrier removal decisions in connectivity impaired river networks to improve the viability of diadromous fish populations. In particular, we integrate spatially explicit population viability analysis (PVA) into an optimization framework to maximize equilibrium population size. A case study involving a wild coho salmon (*Oncorhynchus kisutch*) population from the Tillamook basin along the Oregon Coast, USA is used to illustrate the usefulness of our approach.

Oregon Coast coho salmon were first listed as a threatened species under the Endangered Species Act (ESA) in 1998 by the National Marine Fisheries Service (NMFS) (NMFS, 2015). Although delisted from the ESA in 2001, following a court case, NMFS subsequently relisted the species in 2008 and reaffirmed the listing status again in 2011.

Blocked fish passage has resulted in extensive loss of access to historical coho salmon habitats within estuaries, tidal freshwater, and upstream areas. The resulting loss of longitudinal stream connectivity has reduced the availability of habitat types, negatively affecting species productivity, abundance, spatial structure, and genetic diversity. Improved fish passage is essential to the successful recovery of Oregon Coast coho (NMFS, 2015).

A detailed review of procedures and techniques related to evaluating and prioritizing mitigation of fish passage barriers is presented in Kemp and O'Hanley (2010). Given multiple and often conflicting environmental and economic goals, they recommend the use of optimization models and multi-criteria decision making techniques as an objective and cost-effective means for prioritizing barrier repair and removal decisions. A review of the literature shows that most optimization based approaches for barrier removal aim to maximize the amount of accessible (i.e., connectivity weighted) habitat available to migratory fish. For example, O'Hanley and Tomberlin (2005) devised a nonlinear integer program to optimize the removal of fish passage barriers. The goal of their model is to maximize net gain in accessible habitat for diadromous fish given a limited budget for barrier mitigation. Kuby et al. (2005) propose the use of a bi-objective optimization model for prioritizing the removal of large hydropower dams. The two objectives of the model are to: 1) maximize the amount of reconnected salmon habitat and 2) minimize the loss of hydropower and water storage capacity. O'Hanley (2011) presents an optimization model for prioritizing the removal of artificial passage and flow barriers which negatively affect river ecosystems. The objective is to decide which barriers to remove, subject to a budget, in order to maximize the length of the single largest unobstructed subsection of river. The models by Kuby et al. (2005) and O'Hanley (2011) assume barriers have binary passabilities (i.e., either fully impassable or fully passable). In contrast, O'Hanley and Tomberlin (2005) allow for partial barrier passability.

Ioannidou and O'Hanley (2018) also consider habitat accessibility in their optimization framework. Their multi-objective optimization model, which is mainly based on Chapter 2, aims to maximize the total hydropower production by installing new small hydropower plants (SHPs) while limiting impacts on river connectivity. Their analysis accounts for the interactive effects that SHPs have on hydropower production and fish passability. The model allows for fractional values of barrier passability.

Among the few to consider fish population and dispersal dynamics is a study by Paulsen and Wernstedt (1995). The authors develop a combined simulation and optimization framework to analyze the cost-effectiveness of potential mitigation measures aimed at restoring salmon populations affected by hydropower dams in the Columbia River basin. The simulation model is used to evaluate the biological effects of possible management alternatives. A linear programing model is subsequently employed to find the least-cost solution that satisfies a set of stock harvest and escapement goals. As pointed out in Kemp and O'Hanley (2010), this approach is limited to dealing with a fairly small number of barrier mitigation / habitat restoration alternatives as each feasible combination of alternatives needs to be individually simulated.

Another notable example is Zheng et al. (2009) who propose a mixed integer linear programing model for optimizing dam removals in the Lake Erie basin. The model is multi-objective and aims to maximize a combination of ecological (e.g., native species biomass) and socio-economic (e.g., recreational and commercial harvesting) goals subject to a budget constraint on dam removal and invasive species (i.e., sea lamprey) control costs. Zheng and Hobbs (2013) extend the work of Zheng et al. (2009) by adding a third objective: the risk of dam failure. A mixed integer linear program in conjunction with two cost regression models, three ecological models, and a dam safety assessment tool are used to illustrate trade-offs of dam removal projects in terms of public safety, fish population health and cost in the Lake Erie basin. Both Zheng et al. (2009) and Zheng and Hobbs (2013) make the strong simplifying assumption that changes in fish population sizes are locally linear in response to dam removal.

Newbold and Siikamaki (2009) is another relevant study. In order to prioritize watershed conservation activities, they develop a PVA model for Columbia River salmon and incorporate it in a reserve site selection (RSS) procedure. Their aim is to improve the long-run probability of persistence for salmon. They develop a stochastic population model and combine it with habitat quality models. Watershed protection decisions are prioritized by the RSS model based on their cost-effectiveness. Options for barrier removal are not considered.

Looking beyond optimization based studies, PVA models have seen extensive use in the context of river habitat management. A nice example is the paper by Nieland et al. (2015), which examines mortality impacts of large hydropower dams on an Atlantic salmon population in the Penobscot River, Maine. The model, which tracks both the number and location of fish at multiple life stages, is used to evaluate relative changes in abundance of six dam removal scenarios.

In a related study, Harvey and Railsback (2012) analyze the effects of fish passage barriers on a virtual resident trout population. A detailed individual-based model is developed to capture the demographics and fine-scale movements of trout. Simulations of five scenarios with varying barrier densities are analyzed to investigate how the location of barriers affect two population stability properties: persistence and resistance. Interestingly, they find that low barrier densities can actually produce an increase in overall biomass.

Scheuerell et al. (2006) propose a framework to evaluate the effects of habitat change, hatchery operations, and harvest management actions on salmon population status. They use a multistage Beverton-Holt population model to describe the production of salmon from one life stage to the other and to provide estimates of abundance, productivity, spatial structure, and diversity. Their framework is used to evaluate the potential consequences of habitat conservation alternatives in Snohomish River basin in Washington State.

Nickelson and Lawson (1998) developed a life cycle model to estimate the extinction probability of coho salmon populations along the Oregon Coast. Spawner abundance, demographic and environmental stochasticity, genetic effects, density, and habitat driven survival rates are all taken into account. Simulations are run to evaluate the viability of coho salmon over a 99 year period. A comprehensive review on the use of PVA models in the planning of recovery actions for Atlantic and Pacific salmon can be found in Sweka and Wainwright (2014).

3.2 Materials and Methods

3.2.1 Maximizing Equilibrium Fish Population Size

In what follows, we propose a decision planning tool, referred to as MaxPop, for cost-effectively targeting the mitigation (e.g., removal, replacement, or retrofitting) of in-stream barriers that negatively impact river connectivity. Our aim is to determine which set of barrier mitigation actions will lead to the largest long-term equilibrium population size for a given species of interest via increased dispersal and habitat utilization. In our current application, we restrict ourselves to fish species with a diadromous life-cycle, focusing specifically on salmon.

We make the following assumptions. The river network under consideration is strictly "dendritic," meaning it never diverges in the downstream direction, thus excluding braided river systems. Given this assumption, there is a unique path from the river mouth to any point upstream. We further assume that each barrier has a known passability value. Passability refers to the fraction of fish, in the range 0-1, that can successfully pass a barrier (Kemp and O'Hanley, 2010). Passability is normally species, life-stage (e.g., adult versus juvenile), and directionally (i.e., upstream versus downstream) dependent. Cumulative passability, as it has already been discussed in Chapter 2, refers to the combined effect that barriers have on fish dispersal. Assuming that barrier passabilities are independent, cumulative passability is calculated by multiplying the passabilities of all barriers along the path from a given origin (e.g., the ocean) to a given destination (e.g., an upstream spawning area). Barrier mitigation is carried out to increase the upstream and/or downstream passability of a barrier for one or more life-stages. For any particular barrier, there may be multiple mitigation options available. The total cost of mitigation cannot exceed a predefined budget.

The MaxPop model is formulated as follows. Let J, indexed by j, be the set of physical barriers, both artificial and natural, within a river network, and let J' be the subset of artificial barriers. The set of barrier mitigation actions available at each artificial or natural barrier j is denoted by A_j. A_j is empty when no mitigation options are available at site j. The cost of implementing mitigation option i at barrier j is given by c_{ji}. The total barrier mitigation budget is denoted by b. Decision variable x_{ji} to equal 1 if mitigation option i is selected for barrier j, 0 otherwise. Given a vector of barrier mitigation decisions $\mathbf{x}$, function $F(\mathbf{x})$ expresses the equilibrium population size of a species (e.g., number of breeding adults) based on dispersal behavior (possibly life-stage specific), level of river connectivity, and population growth dynamics.

$$\max_{\mathbf{x}} F(\mathbf{x}) \tag{3.1}$$

s.t.

$$\sum_{j \in J'} \sum_{i \in A_j} c_{ji} x_{ji} \leq b \tag{3.2}$$

$$\sum_{i \in A_j} x_{ji} \leq 1 \qquad \forall j \in J' \tag{3.3}$$

$$x_{ji} \in \{0, 1\} \qquad \forall j \in J', i \in A_j \tag{3.4}$$

The objective (3.1) is to maximize equilibrium population size. Constraint (3.2) requires that the total cost

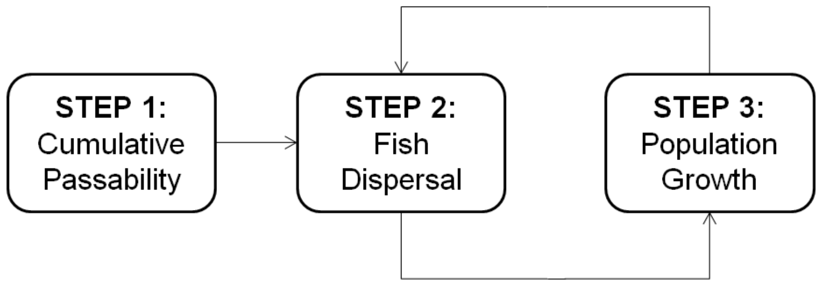


Figure 3.1: Steps in determining equilibrium population size.

of implementing the barrier mitigation actions does not exceed the available budget. Constraints (3.3) ensure that at most one mitigation project can be carried out at each artificial barrier. Finally, constraints (3.4) impose binary restrictions on the barrier mitigation decision variables.

The three steps in determining the long-term population size $F(\mathbf{x})$ corresponding to barrier mitigation solution $\mathbf{x}$ are outlined in Figure 3.1.

Step 1

The cumulative passability to spawning/rearing areas is determined based on the barrier passabilities specified by solution $\mathbf{x}$. For spawning adult salmon, cumulative passability α_j to spawning areas immediately above barrier j would be calculated as $\alpha_j = \prod_{k \epsilon D_j} \left(p_k^0 + \sum_{i \in A_j} p'_{ji} x_{ji} \right)$, where D_j is the subset of barriers downstream from and including barrier j, p_j^0 is the initial passability of barrier j, and p'_{ji} is the increase in passability given implementation of mitigation option i at barrier j.

Step 2

Fish dispersal to spawning/rearing occurs according to the type of adult/juvenile dispersal pattern and level of connectivity.

Step 3

Population growth takes place in this step, with the number of recruits being produced in a particular habitat area possibly density dependent.

Note that Steps 2 and 3 can be formed of multiple sub-steps if and when dispersal and productivity/survival are life-stage dependent. Steps 2 and 3 need to be repeated iteratively from one generation to the next until equilibrium population size is achieved. Depending on the type of homing pattern, fish dispersal (Step 2) may need to be recalculated at each generation. Each barrier mitigation solution $\mathbf{x}$ will normally result in a different equilibrium population size, which means that the whole process needs to be repeated any time a new barrier mitigation solution is evaluated.

In what follows, the various homing patterns considered in our study are discussed in Section 3.2.2. In Section 3.2.3, we cover fish population growth dynamics. Our solution methodology is presented in Section 3.2.4. Finally, in Section 3.2.5 we provide background information about our study area and the input data used to parameterize our model.

3.2.2 Dispersal Patterns

Upstream migrating adult salmon tend to return (i.e., home) to their natal river locations to spawn. Salmon homing can be conceptualized along a hierarchy of spatial scales (Quinn, 1997), starting from the river basin, followed by main tributary, then stream reach, and finally down to a specific point of a stream reach (i.e., the redd). Naturally, homing is more accurate at broader spatial scales. Usually a small percentage of adult fish, referred to as "strays," move into non-natal streams during upstream movement, which has implications for metapopulation persistence. In this study, two extreme homing patterns are considered: river and reach homing. The first type of homing behavior (river homing) assumes that adults have low homing fidelity; adults will return to their natal river and then disperse freely within the river to find suitable spawning habitat. For the second type of homing behavior (reach homing), it is assumed that adults have much higher homing fidelity and will attempt to return to their specific natal stream reach. These two dispersal patterns are discussed in detail below. For simplicity, juvenile fish are assumed to have suitable rearing habitat within the vicinity of the spawning area from which they emerge and so do not make appreciably long distance dispersal movements to upstream/downstream rearing areas.

3.2.2.1 River Homing

With river homing, adult salmon are assumed to distribute within their natal river according to an ideal free distribution (IFD) (Case, 1999). Under IFD, "consumers" (i.e., fish), have ideal knowledge of habitat resources and disperse in such a way that the density of consumers is uniform. In this study, habitat resources are assumed to be proportional to river length. If no barriers were present, the number of spawners per unit length of river would be the same in each spawning area. With barriers present, however, dispersal is disrupted and equal densities cannot necessarily be achieved.

To model the impact that barriers have on pushing spawner densities away from what would be expected based on IFD, we develop a linear program (LP) referred to as the Ideal Free Distribution with Barriers Problem (IFDBP). The model seeks to minimize the maximum difference between an ideal spawner density and what can be achieved given the presence of barriers. In addition to the notation introduced previously, let p_j be the current passability of barrier j and let D_j be the subset of barriers downstream from and including barrier j. Following King and O'Hanley (2016), the section of river above a barrier up to the next set of barriers or the river terminus is referred to as a river subnetwork. The amount of spawning habitat in subnetwork j is given by v_j, while the total amount of spawning habitat within the river catchment is denoted by V (i.e., $V = \sum_{j \epsilon J} v_j$). The total number of spawners is N. Finally, consider the following decision variables.

$y_j =$ number of spawners dispersing to subnetwork j

$z =$ maximum difference between observed and ideal spawner density for any subnetwork

With this in place, a mathematical formulation of IFDBP is given below.

$$\min z \tag{3.5}$$

s.t.

$$\sum_{j \epsilon J} y_j = N \tag{3.6}$$

$$y_j \leq \prod_{k \epsilon D_j} p_k N \qquad \forall j \in J \tag{3.7}$$

$$z \geq \frac{1}{v_j} y_j - \frac{1}{V} N \qquad \forall j \in J \mid v_j > 0 \tag{3.8}$$

$$y_j = 0 \qquad \forall j \in J \mid v_j = 0 \tag{3.9}$$

$$y_j \geq 0 \qquad \forall j \in J \tag{3.10}$$

The objective function (3.5) minimizes the largest difference in spawner density from ideal across all river subnetworks. The first constraint (3.6) forces the sum of the spawners across all subnetworks $\sum_{j \epsilon J} y_j$ to be equal to the total number of spawners N. Constraints (3.7) restrict y_j not to exceed the total number of fish that can potentially reach subnetwork j, which is equal to the cumulative passability $\prod_{k \epsilon D_j} p_k$ of barrier j times the number of spawners N. Constraints (3.8) specify for all subnetworks j with non-zero spawning habitat ($v_j > 0$) that the maximum difference in spawner density z from ideal must be greater than or equal to the observed density y_j/v_j in subnetwork j minus the ideal density N/V. Inequalities (3.9) force y_j to be zero for all subnetworks with zero spawning habitat length, since no fish would migrate to such areas. Lastly, inequalities (3.10) impose non-negativity restrictions on the y_j decision variables.

Note that when evaluating a salmon metapopulation that distributes among multiple rivers, model IFDBP needs to be solved separately for each river network. Before doing so, the number of outgoing strays and the number of incoming straying needs to be taken into account in order to specify the correct spawner population sizes N that will distribute within each river network.

Further note that as long as cumulative passability values do not change (i.e., no additional mitigation is carried out) then the relative proportions of fish migrating to any particular river subnetwork will remain constant even if the total spawner population size N subsequently changes in later generations. The importance of this is that IFDBP only needs to be solved once when determining the equilibrium population size.

3.2.2.2 Reach Homing

The second adult dispersal pattern examined in this study is reach homing. Here, fish are expected to return to the locality of the natal stream reach from which they emerged, which for our purposes is taken to be their originating river subnetwork. If no barriers were present, then any fish spawned in subnetwork j would be able to return to j as adults. With barriers present, however, of the number of fish N^j spawned in subnetwork j, only a fraction, equal to N^j times the cumulative passability $\prod_{k \epsilon D_j} p_k$ of barrier j, will be able to do so. The rest will be "trapped" in the subnetworks downstream of j.

Consequently, with reach homing, the number of spawners y_j contained in subnetwork j will be the sum of the spawners that originated there and successfully returned plus a portion of spawners that were unsuccessful in reaching subnetworks further upstream due to passability restrictions. In addition, the number of spawners within subnetwork j will be affected by fish straying. In particular, a small percentage of fish spawned in j will stray away from j to other subnetworks and a small number of fish will be redirected to subnetwork j after straying from other reaches. For simplicity, we assumed a fixed percentage of fish would stray from any subnetwork and then redistribute themselves by spreading equally among other river catchments and then with equal probability among subnetworks in each river catchment. Unlike with river homing, fish dispersal calculations need to be updated for every generation for reach level homing.

3.2.3 Population Growth

Density dependent population growth is a well-established principle (Wainwright et al., 2008). No population can grow indefinitely, so the growth rate will approach 1, once a population approaches the limits of its resources (Morris and Doak, 2002). The deterministic population model used in MaxPop is the Ricker model, which is widely used in fish population viability analysis when density dependent population growth is assumed. According to the Ricker model, the expected number of individuals in any generation $t+1$ is a function of the number of individuals in the previous generation t. More precisely, given the number of spawners N_t in generation t, the number of offspring (recruits) N_{t+1} in the next generation is given by:

$$N_{t+1} = N_t e^{r\left(1-\frac{N_t}{K}\right)} \tag{3.11}$$

where r is the intrinsic growth rate and K is the carrying capacity of the habitat. When r is greater than 0 and less than 2 (i.e., $0 < r < 2$), the model has a stable equilibrium. Cycles or chaotic dynamics are produced for growth rates $r \geq 2$ (Morris and Doak, 2002).

In our implementation of MaxPop, given an initial population size N_0^j in each subnetwork j, the Ricker model is used to generate the number of recruits produced in each subnetwork over 200 generations. Recruits produced in each subnetwork first travel back to the sea and then disperse upstream according to one of the aforementioned dispersal patterns in order to produce the next generation. The mean of the last 100 generations is used to compute the "equilibrium" population size, which may be stable, cyclic, or chaotic (Morris and Doak, 2002).

3.2.4 Solution Methodology

A flow chart of the heuristic algorithm used to solve MaxPop is shown in Figure 3.2. In Step 1, an initial starting solution is generated by solving a standard barrier optimization model, referred to as MaxHab, which maximizes the amount of accessible (i.e., connectivity-weighted) habitat within a river catchment available to upstream migrating fish (O'Hanley and Tomberlin, 2005; King and O'Hanley, 2016). The equilibrium population size of this solution is computed (taking into account dispersal and population dynamics) and then this solution is accepted as the current best (aka incumbent) solution.

For clarity, using the notation introduced earlier and the following decision variables

a_j = cumulative passability to river habitat immediately above location j

we present the nonlinear formulation of the MaxHab model:

$$\max \sum_{j \epsilon J} a_j v_j \tag{3.12}$$

s.t.

$$a_j = \prod_{k \epsilon D_j} (p_k^0 + \sum_{i \epsilon A_k} p_{ik}^{'} x_{ik}) \tag{3.13}$$

$$\sum_{j \in J'} \sum_{i \in A_j} c_{ji} x_{ji} \leq b \tag{3.14}$$

$$\sum_{i \in A_j} x_{ji} \leq 1 \qquad \forall j \in J' \tag{3.15}$$

$$x_{ji} \in \{0, 1\} \qquad \forall j \in J', i \in A_j \tag{3.16}$$

In Step 2, a local search is performed in an attempt to find a solution with higher equilibrium population size. Here, a currently mitigated barrier is selected and its passability and the passabilities of all other mitigated barriers upstream from the selected barrier are temporarily reset to their initial passability values. The resulting cost savings from undoing mitigation for the selected barrier and those upstream is added back to the remaining budget and the equilibrium population size is recalculated.

A new candidate solution is then constructed using a "greedy" add procedure, whereby the barrier mitigation option with the largest benefit-to-cost ratio (net change in equilibrium population size divided by cost) is iteratively selected until either the remaining budget is exhausted or no improvement in equilibrium population size can be achieved. In order to estimate the equilibrium population size for each candidate solution the three step procedure described in Section 3.2.1 should be followed. Note that when considering any given barrier for mitigation, if zero-passability barriers are present downstream, these are all mitigated at the same time. Intuitively, it would never make sense to mitigate a barrier if cumulative passability were to remain zero due to the presence of impassable barriers downstream.

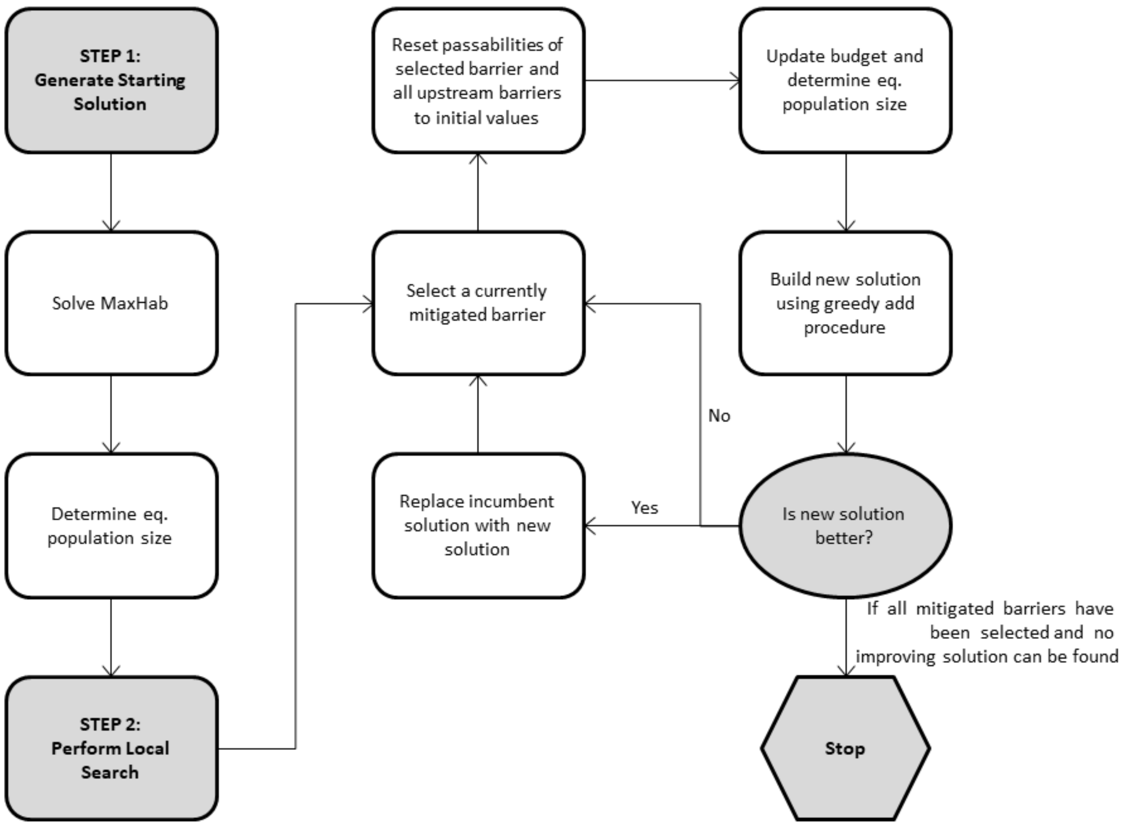


Figure 3.2: MaxPop solution algorithm.

Once a proposed candidate solution has been built, it is then compared to the incumbent solution. If the candidate solution results in a higher equilibrium population size, it replaces the incumbent solution. Otherwise, the algorithm goes back to the incumbent solution and the above procedure is repeated for a different selected barrier.

3.2.5 Study Area

A case study exploring the population and dispersal dynamics of coho salmon (*O. kisutch*) in the Tillamook basin, Oregon, USA is used to illustrate the benefits of our proposed modeling framework. Our barrier dataset, consisting of the location of 202 culverts, dams, fords, and tidegates, is derived from Pilson (2012). Each barrier is georeferenced and includes a description of the type of structure, available mitigation options, and estimated costs. Initial passability values for each type of barrier are shown in Table 3.1.

In order to account for potential habitat between the river mouth and the first set of artificial barriers, it was necessary to add "dummy" (i.e., fully passable) barriers at each river mouth. Working off a 1:100,000 scale river network layer created by the Oregon Department of Forestry, mouth nodes were identified using the RivEX toolbox for ArcGIS 10.2.1 and were added to our barrier dataset with no available mitigation projects and initial passability equal to 1. RivEX was also used to determine key barrier metrics, such as each barrier's immediate downstream barrier (from which set D_j could be constructed) and net upstream river length (from which parameter v_j could be determined). Barrier points were snapped to the river network using a 50 m snapping distance. This resulted in a final dataset with 193 barriers and 19 mouth nodes spread among 6 watersheds (the Miami, Kilchis, Tillamook, Trask, Wilson and Tillamook Bay watersheds), as shown in Figure 3.3.

Coho salmon usually spawn in small streams (NOAA, 2017), so for our case study we considered as spawning habitat all river segments with Strahler stream order 1. RivEX was used to extract the Strahler steam order for each river segment. At present (i.e., prior to any barrier mitigation), the total length of accessible spawning habitat is estimated to be 407.73 km within the Tillamook basin.

Table 3.1: Initial passability values for Tillamook Bay barriers.

Barrier Type	Passability
Culvert	0.2
Dam	0
Ford	0.9
Tidegate	0.9
Weir	0
Other	0

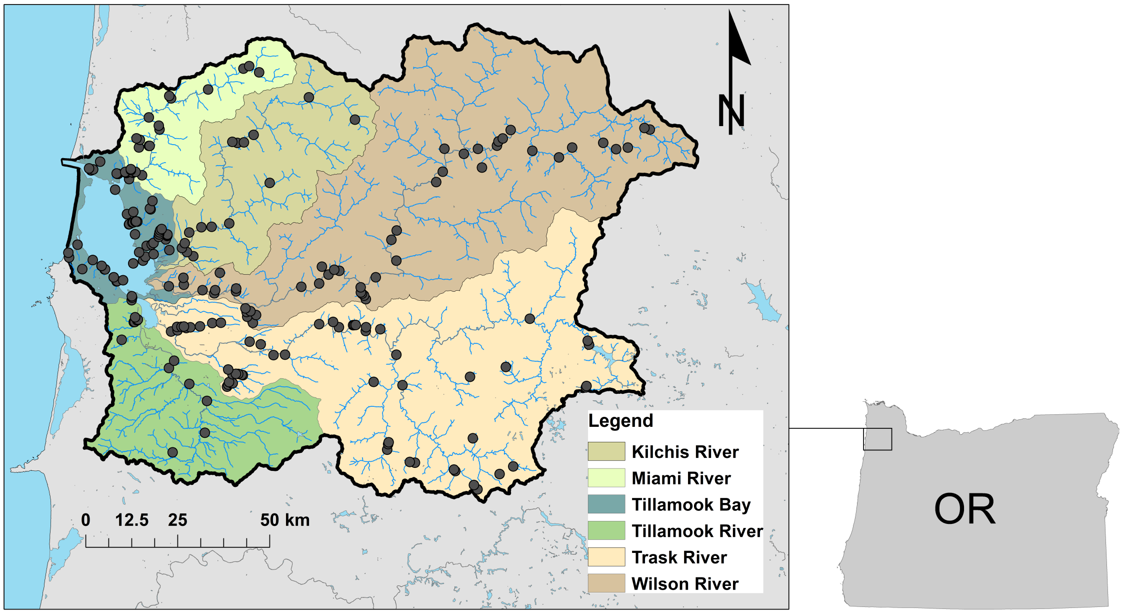


Figure 3.3: Watersheds and barrier locations (dark grey circles) in Tillamook basin.

Population counts and harvest rates for wild coho spawners in the Tillamook basin for the period 1996 to 2013 were obtained from the Oregon Adult Salmonid Inventory and Sampling Project (OASIS, 2016). Recruits were assumed to return as adults 3 years after hatching, with the number of recruits produced in year t equal to $N_{t+3}/(1-h)$, where N_{t+3} is the number of recorded spawners 3 years after time t and h is the harvest rate. A plot of the estimated numbers of Tillamook coho spawners versus recruits and the curve for the fitted Ricker spawner-recruitment model are shown in Figure 3.4. Ricker model parameters were estimated using simple linear regression. The regression model, which had an adjusted R^2 of 0.725, produced estimates of $r = 1.70$ and $K = 8442$ (overall carrying capacity in the Tillamook basin). By comparison, the adjusted R^2 for a Beverton-Holt model was 0.612, indicating that the Ricker model is a better choice for describing density-dependent growth of wild Tillamook coho. Carrying capacity was subsequently translated to $K = 16$ spawners per river km based on currently accessible spawning habitat. The straying rate was set at 3%, a mid range value for wild coho salmon (Labelle, 1992).

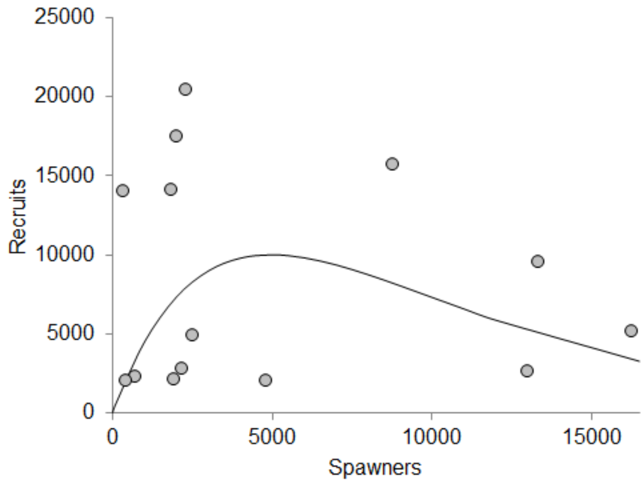


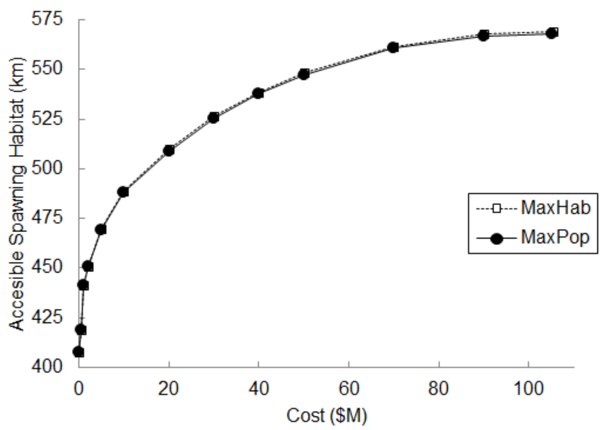
Figure 3.4: Estimated spawners and recruits (gray circles) for Tillamook basin wild coho (*Oncorhynchus kisutch*) and fitted Ricker spawner-recruitment model (solid black curve).

3.3 Results

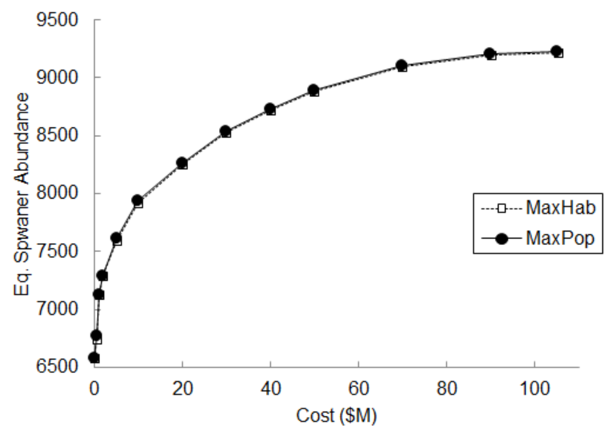
Results for the MaxPop model are presented in Figures 3.5 and 3.6. Accessible habitat and equilibrium population size are plotted against cost of barrier mitigation for both reach and river homing dispersal patterns. For comparison purposes, results are also reported for model MaxHab, which maximizes accessible habitat.

According to our findings, homing behavior has a significant impact on optimal barrier mitigation strategies. With reach homing (Figure 3.5), MaxPop more or less produces the exact same levels of accessible habitat and population size as MaxHab. Under this dispersal pattern, mitigation actions that maximize accessible habitat also maximize spawner abundance. The sets of barriers selected for mitigation are nearly identical for both models across all budget scenarios considered.

With river homing (Figure 3.6), however, MaxPop and MaxHab produce very different mitigation strategies. In most cases, MaxPop achieves a given population size target by removing far fewer barriers, and hence at much lower cost, than MaxHab. For example to reach a population size of 9000 spawners, 85 barriers would need to be removed at a cost of roughly \$20M according to MaxHab. In contrast, MaxPop is able to achieve a comparable spawner abundance (8962) by removing only 7 barriers at a cost of around \$2M (a 90% cost savings). Similarly, to reach a maximum of 9221 spawners, MaxPop recommends the removal of 37 barriers at a cost of \$14.7M, while MaxHab only achieves a similar population size (9217) when 166 barriers are removed at a cost of nearly \$70M (a 79% cost savings). MaxPop, that focuses on improving population sizes instead of maximizing habitat, succeeds to reach high population abundance by suggesting much cheaper solutions.

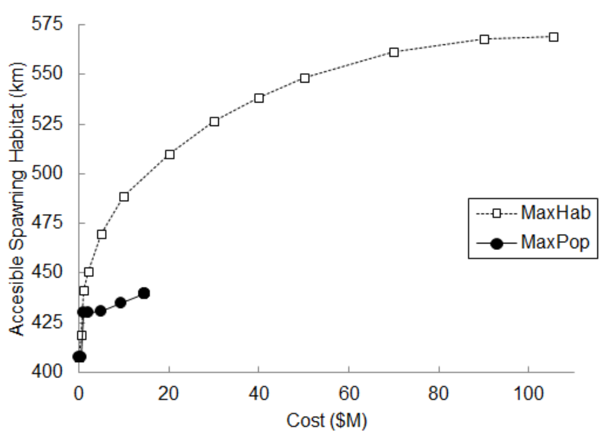


(a)

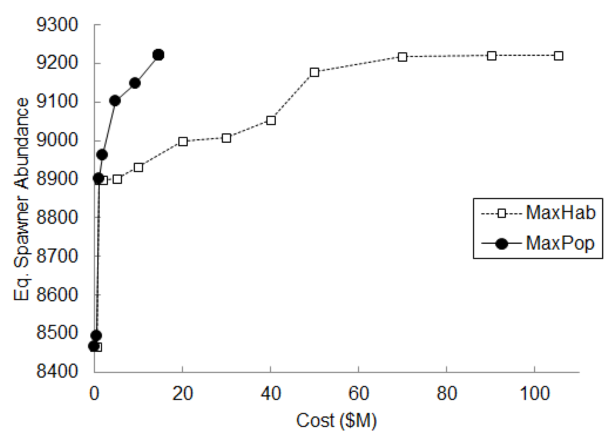


(b)

Figure 3.5: Accessible spawning habitat (a) and equilibrium spawner abundance (b) versus barrier mitigation cost for MaxPop and MaxHab given reach homing.



(a)



(b)

Figure 3.6: Accessible spawning habitat (a) and equilibrium spawner abundance (b) versus barrier mitigation cost for MaxPop and MaxHab given river homing.

On the other hand, it is certainly true that accessible habitat is substantially lower for MaxPop than MaxHab under a river homing dispersal pattern. For example, accessible habitat for MaxHab eventually reaches a maximum of 569 km of accessible habitat at a cost of a little over $105M. MaxPop, however, only ever goes up to 440 km of accessible habitat (23% less) at the highest cost level. This shows that there is not necessarily a need to maximize accessible habitat if one aims to maximize equilibrium population size. If the aim is to reach the highest possible level of total accessible habitat MaxHab would be the right choice.

It is also worth pointing out that the different dispersal patterns lead to substantially different estimates of equilibrium population for MaxPop. In particular, with reach homing, population sizes range from 6580 (current) to 9229 (maximum) spawners. Here, barrier removal yields large gains (almost 40%) in fish abundance. For river homing, however, gains in fish numbers are much more modest, going from 8467 to 9221 (a 9% increase).

To account for the effects of environmental stochasticity on fish population growth, an extension of MaxPop, referred to as MinExP, was developed that seeks to minimize the probability of population extinction over a given time horizon. More specifically, environmental variation is introduced by replacing the deterministic equation for population growth (3.11) with the following:

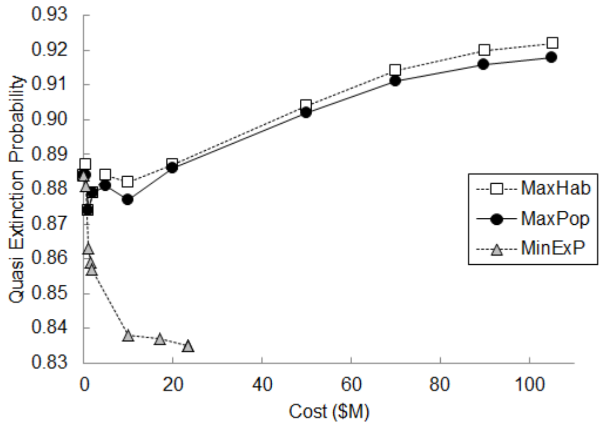
$$N_{t+1} = N_t e^{r\left(1-\frac{N_t}{K}\right)+\varepsilon_t} \tag{3.17}$$

Parameter ε_t, which adjusts the underlying growth rate $r\,(1 - N_t/K)$ up or down, is drawn from a normal distribution and has a mean of zero and a variance of $nV_r/\,(n-1)$, where n is the number of data points used in the linear regression for the Ricker model and V_r is the residual variance (Morris and Doak, 2002).

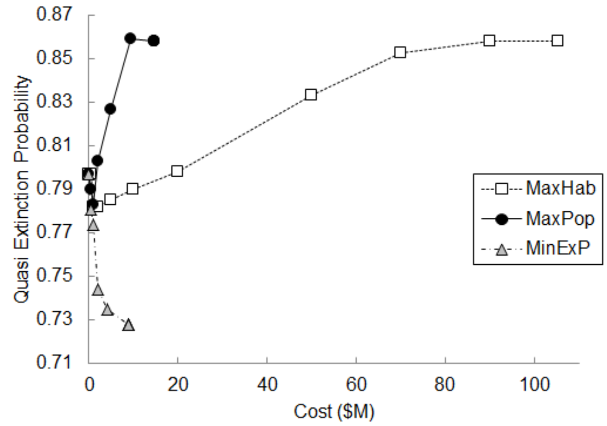
To yield estimates for the probability of extinction, population sizes were simulated across 50 generations using equation (3.17) and the fraction of simulation runs (out of 1000) in which population abundance fell below a quasi-extinction threshold (QET) was determined. As in Newbold and Siikamaki (2009), we used a QET of 10% of recent (1996-2013) average abundance for wild Tillamook coho salmon, which equates to 615 spawners.

Modified Heuristic

To solve MinExP, we used the same basic heuristic method applied to MaxPop but with a few modifications. In particular, a) we changed the objective from maximizing equilibrium population size to minimizing quasi-extinction and b) in the case of river homing, we used the solutions produced by MaxPop as the initial starting solutions instead of the MaxHab solutions.



(a)



(b)

Figure 3.7: Quasi-extinction probabilities versus barrier mitigation cost for MinExP, MaxPop, and MaxHab given reach homing (a) and river homing (b).

Estimated quasi-extinction probabilities for solutions to MinExP and MaxPop are provided in Figure 3.7. In all cases, 95% confidence intervals were within ±2.8% of the reported mean extinction probability.

The main observation from Figure 3.7 is the much lower extinction risk achieved by MinExP for any given level of cost in comparison to MaxPop or MaxHab, indicating that additional benefits can be gained by incorporating environmental stochasticity. Without any mitigation action being undertaken, the probability of the Tillamook coho population reaching the quasi-extinction threshold in 50 generations (~150 years) is 88.4% under reach homing and 79.9% under river homing. For MinExP, extinction probabilities rapidly decreases as barrier mitigation resources increase, eventually reaching a minimum of 83.5% for reach homing at a cost of $23.2M and 72.8% for river homing at a cost of just $8.8M. Interestingly, MinExP achieves these minimum extinction probabilities by removing only a small subset of barriers - just 43 barriers for reach homing and 18 barriers for river homing that are mostly concentrated low in the catchment, as shown in Figure 3.8.

In comparison, extinction risk initially goes down for both MaxPop and MaxHab but then goes up as barrier mitigation resources and the number of barriers removed increase, eventually rising above current quasi-extinction probabilities. This occurs regardless of the type of homing behavior. When mitigation resources are unrestricted, MaxPop suggests the removal of 191 barriers at a cost of $105.1M for reach homing and the removal of 37 barriers at cost $14.7M for river homing (see Figure 3.8). Quasi-extinction risk is not only much higher than MinExP (+8.3% for reach homing and +13.0% for river homing), but also higher than under a no-mitigation scenario (+3.4% for reach homing and +6.2% for river homing). Removing all 193 barriers at a cost $105.3M, as recommended by MaxHab, similarly would cause extinction risk to go up by +3.8% under reach homing and by +6.1% under river homing.

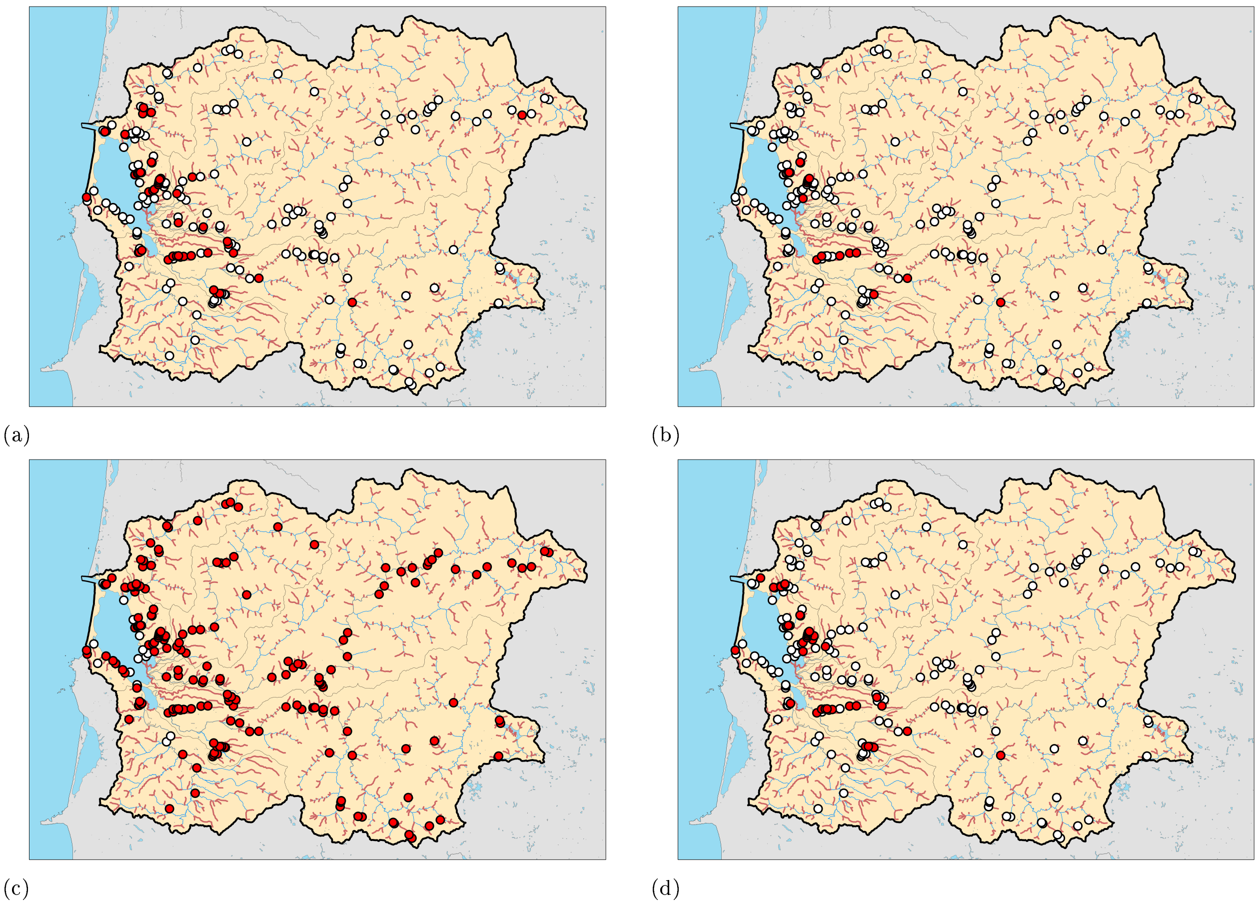


Figure 3.8: Barriers targeted for mitigation in the Tillamook basin by MinExP for reach homing (a) and river homing (b) and by MaxPop for reach homing (c) and river homing (d) when mitigation resources are unrestricted. Selected barriers are represented by red circles, unselected barriers by white circles. Spawning areas are indicated by bold, pale red colored river segments.

Homing pattern appears to have moderate influence on extinction risk. Looking just at MinExP, extinction probabilities are strictly lower for river homing, ranging from a high of 79.7% to a low of 72.8%. Under reach homing pattern, extinction probabilities reach a high of 88.4% (+8.7%) and a low of 83.5% (+10.7%).

We emphasize that the results reported here are only hypothetical. Firstly, they are based a specific form of density dependence, namely the Ricker model. Different models for density dependence (e.g., Beverton-Holt, Gompertz, hockey stick) may produce very different outcomes. Second, we ignore various other factors in our analysis that may be important to long-term population growth and viability, such as juvenile dispersal and survival, Allee effect, the effects of pollution on habitat quality, demographic stochasticity, hatchery operations, and straying from coho salmon populations outside the Tillamook.

3.4 Discussion

The main goal of this study was to explore how optimal barrier mitigation strategies are affected by the consideration of fish dispersal and population dynamics. Based on our case study of a wild coho salmon (*O. kisutch*) population from the Tillamook basin, we find that the choice of homing pattern for spawning adults has a very large influence in determining which barriers should be mitigated to maximize equilibrium abundance. With reach homing, essentially the same equilibrium population sizes are achieved by models MaxPop and MaxHab, meaning that maximizing accessible habitat in effect also maximizes population size. In short, there does not appear to be much benefit from using the more complex and computationally expensive MaxPop model.

With the river homing, however, this is decidedly not the case. For most budget levels, solutions to MaxPop differed markedly from MaxHab. In particular, MaxPop recommends the removal of a much smaller number of barriers in order to maximize spawner population size. What this suggests is that focusing on maximizing accessible habitat may lead to the removal of an excessive numbers of barriers at high cost, while yielding relatively little in terms of increased fish population size. Indeed, using MaxPop to maximize equilibrium population size for Tillamook coho salmon, assuming a river homing pattern, only requires the removal of 37 out of 193 barriers at a cost of \$14.7M. By using MaxHab to maximize accessible habitat, 166 barriers at a cost of \$70M would need to be removed to achieve roughly the same equilibrium population size. The mitigation of the 166 barriers selected by MaxHab would achieve the maximum accessible habitat given a \$70M but would not improve the maximum equilibrium population size already achieved by mitigating the 37 barriers selected by MaxPop. In other words spending an extra of \$55.3M to mitigate 129 additional barriers would only improve total accessible habitat and not population size. MaxPop is the ideal choice if one aims to improve fish population abundance by barrier mitigation.

The inclusion of environmental stochasticity in our analysis also produced some very interesting results. Surprisingly, removing all barriers resulted in higher quasi-extinction risk compared to leaving all existing barriers in place regardless of homing pattern. According to model MinExP, the probability of quasi-extinction within 50 generations without any mitigation action being implemented is 88.4% for reach homing and 79.7% for river homing. Extinction probability increases to 92.2% (reach homing) and 85.8% (river homing) when all 193 barriers are removed. Under a river homing dispersal pattern, the lowest extinction risk that could be achieved was 72.8% via the removal of 18 barriers at a cost of $8.8M. For reach homing, the lowest extinction risk was 83.5% and was achieved by removing 43 barriers at a cost of $23.2M.

The most straightforward explanation for this is a population thinning effect caused by the presence of fish passage barriers under density dependent population growth. More specifically, depending on the spatial distribution of barriers and spawning habitat, limited amounts of river fragmentation can depress spawner densities in certain reaches/subnetworks below carrying capacity, thereby allowing a surplus of recruits to be produced. This, in turn, can help to artificially boost population numbers and improve population persistence in a manner similar to how limited harvesting can potentially increase population growth vis-à-vis the maximum sustainable yield principle (Case, 1999). We emphasize that our results are theoretical, but are supported by Harvey and Railsback (2012) who also observed that the largest abundance for a virtual resident trout population occurred at low but positive barrier densities.

In our current study, we focused on adult salmon dispersal, while assuming that juveniles are able to access rearing habitats near to where they emerged. An interesting extension of our work would be to consider a stage-structured population model and examine how habitat needs and dispersal dynamics at each life stage interact with barrier mitigation decisions. Another avenue for future research would to be embed barrier removal planning within a wider ecosystem level approach that considers the role of interspecific competition, predator-prey, and parasite-host dynamics.

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Chapter 4

The Hidden Elephant in the Room: Large-Scale River Connectivity Restoration Requires Planning for the Presence of Unrecorded Barriers

Habitat fragmentation is a leading threat to global biodiversity. Restoring habitat connectivity, especially in freshwater systems, is considered essential in improving ecosystem function and health. Various studies have looked at cost effectively prioritizing river barrier mitigation decisions. In none of these, however, has the importance of accounting for the potential presence of unknown or "hidden" barriers been considered. In this study, we propose a novel optimization based approach that accounts for hidden barrier uncertainty in river connectivity restoration planning and apply it in a case study of the US state of Maine. We find that ignoring hidden barriers leads to a dramatic reduction in anticipated accessible habitat gains. Using a conventional prioritization approach, habitat gains are on average 60% lower than expected across a range of budgets when there are just 10% additional but unknown barriers. More importantly our results show that anticipating for hidden barriers can improve potential gains in accessible habitat in excess of 110% when the budget is low and the number of hidden barriers comparatively large. Finally, we find that solutions optimized for an intermediate number of unknown barriers perform well regardless of the actual number of hidden barriers. In other words, we can build-in robustness into the barrier removal planning framework.

Dealing with the hidden elephant in the room could lead to a far more realistic approach of the habitat connectivity restoration issue.

4.1 Introduction

Landscape connectivity is crucial for biological conservation (Fischer and Lindenmayer, 2007; Fahrig, 2003) and is especially critical for the integrity of aquatic ecosystems (Lucas and Baras, 2001). Freshwater lotic systems are particularly vulnerable to barrier fragmentation due to the dendritic structure of river networks (O'Hanley and Tomberlin, 2005; Kemp and O'Hanley, 2010). Large rivers worldwide have been heavily impacted by the construction of river infrastructure (e.g., dams and road crossings) (Bednarek, 2001). In the United States alone, there is an estimated 78,000 dams greater than 3m tall and as many as 3 to 8 million smaller man-made structures that affect natural river flow (Doyle and Havlick, 2009).

Restoration of river connectivity through dam removal and other barrier mitigation actions is universally considered an integral strategy for improving the ecological status of freshwater systems (O'Hanley and Tomberlin, 2005; Bednarek, 2001; Roni et al., 2002). Millions of dollars are spent annually in the US alone on connectivity restoration (Bernhardt et al., 2005; Roni et al., 2008).

Various methods have been suggested to prioritize river barrier mitigation decisions. The more standard prioritization approaches target to improve passage for migratory fish populations (Paulsen and Wernstedt, 1995; O'Hanley and Tomberlin, 2005; Kuby et al., 2005; Neeson et al., 2015; Ioannidou and O'Hanley, 2018; King and O'Hanley, 2016) while fewer studies concentrate on the dispersal of resident fish (O'Hanley, 2011; Cote et al., 2009; Diebel et al., 2010; O'Hanley et al., 2013b). However, none of these studies handles any uncertainty regarding the number or the location of unknown barriers. In practice though, barrier inventories are never exhaustive in recording all potential obstacles that impede fish movements. In the US state of Oregon, for example, around $8,900$ structures were officially recorded in 2004. This number subsequently grew to over $28,000$ by 2011 and to nearly $40,000$ in 2016 (Oregon Department of Fish and Wildlife, 2016). The potential presence of unrecorded or "hidden" barriers raises a key question: what impact does this have on the effectiveness of large-scale connectivity restoration?

In order to answer this question we developed a novel optimization based approach that accounts for hidden barrier uncertainty. Our model identifies the portfolio of barrier removal projects that maximizes the total length of reconnected habitat accessible to migratory fish for a given budget while considering the effects of hidden barriers on the effective habitat length and on the ability of fish to move upstream of known barriers. Barrier passability represents the proportion of fish able to pass a barrier while accessibility refers to the ability of fish to pass all barriers, both known and unknown, from the river mouth to habitat immediately upstream of a barrier. Individual barrier passabilities are assumed to be independent and hidden barriers are assumed to be uniformly distributed throughout the river network.

We applied our optimization approach in the US state of Maine. There is growing interest in reconnecting the heavily disrupted habitat in Maine and especially in restoring the critical habitat for endangered Atlantic salmon (Maine Fish and Wildlife Conservation Office, 2008). Millions are invested annually in the Maine Aquatic Connectivity Project (Natural Resources Conservation Service USDA, 2016) targeting to restore some of the state's highest value aquatic networks.

In our analysis, we investigated the value of factoring in hidden barrier uncertainty in the barrier mitigation planning. We explored the impact that the presence of hidden barriers has on the anticipated gains of uninformed (i.e., that ignore the existence of hidden barriers) barrier mitigation strategies. The solutions identified by our approach were compared against the solutions of a more "standard" prioritization method that ignores hidden barrier uncertainty. The potential gains in accessible habitat of our informed approach have been identified for various hidden barriers scenarios and at a range of budget levels. We also performed a sensitivity analysis to investigate how well solutions optimized for a number of hidden barriers perform when the actual number of hidden barriers is different from the expected.

4.2 Methods

4.2.1 Optimization Model

Our proposed optimization model selects, for a given budget, the set of barriers that should be removed in order to maximize total accessible habitat while taking into account the effects of hidden barriers.

The river network is assumed to have a strictly dendritic structure, meaning that it never diverges in the downstream direction. Assigned to each barrier is a passability score that describes the fraction of fish that are able to pass upstream past a barrier, with 0 denoting a completely impassable structure and 1 a completely passable one. Barrier passabilities are assumed to be independent. As it has been discussed in the previous chapters, cumulative passability represents the combined effect that the barriers have on the fish migration from the river mouth to habitat areas immediately upstream of a barrier and it is evaluated by multiplying the passability of a barrier with the passabilities of all downstream barriers.

Our optimization model extends the one presented in (O'Hanley and Tomberlin, 2005) in order to consider the correlated effects of hidden barriers on both cumulative passability and expected accessible habitat immediately upstream of known barriers.

It is assumed that hidden barriers are uniformly distributed throughout the river network. The expected passability of a hidden barrier is assumed to be equal to the median passability of known barriers.

Using the following decision variables:

$$x_{ji} = \begin{cases} 1 & \text{if mitigation project } i \text{ is carried out at barrier } j \\ 0 & \text{otherwise} \end{cases}$$

$$z_j = \text{cumulative passability (aka accessibility) to habitat area immediately above barrier } j$$

the nonlinear formulation of the hidden barriers removal problem is given as follows:

$$\max \sum_{j \in J} \tilde{v}_j z_j \tag{4.1}$$

$s.t.$

$$\sum_{j \in J^*} \sum_{i \in A_j} c_{ji} x_{ji} \leq b \tag{4.2}$$

$$\sum_{i \in A_j} x_{ji} \leq 1 \qquad \forall j \in J^* \tag{4.3}$$

$$z_j = \prod_{k \in D_j} \left(p_k^0 + \sum_{i \in A_k} p_{ki} x_{ki} \right) \qquad \forall j \in J \tag{4.4}$$

$$x_{ji} \in \{0, 1\} \qquad \forall j \in J^*, \forall i \in A_j \tag{4.5}$$

Here, J is the set of all known artificial and natural barriers. Included in J is a dummy barrier with passability equal to 1 which is used to capture all available habitat between each river mouth and the first set of known barriers. The subset of known artificial barriers is denoted by J^*, D_j is the subset of known artificial/natural barriers downstream from and including barrier j, A_j is the set of mitigation projects available at barrier j, indexed by i, $\tilde{v}_j$ is the expected amount of accessible habitat immediately above barrier j after taking the effects of hidden barriers into account, c_{ji} is the cost of implementing mitigation project i at barrier j, b is the available budget for carrying out mitigation projects, p_j^0 is the initial passability of barrier j and p_{ji} is the increase in passability at barrier j given implementation of mitigation project i.

The objective function (4.1) maximizes total expected accessible habitat. Inequality (4.2) is a budget constraint on the total cost of barrier mitigation. Constraints (4.3) specify that at most one mitigation project can be implemented at each artificial barrier $j \in J^*$. Equations (4.4) determine the cumulative passability of each barrier j (i.e., the product of barrier passabilities in set D_j). Overall passability for any barrier $k \in D_j$ is determined by taking initial passability p_k^0 and adding to it the increase in passability p_{ki} if mitigation project i is selected (x_{ki}). Finally, constraints (4.5) force the barrier mitigation decision variables to be binary.

Note that the equations for cumulative passability (4.4) are nonlinear but can be expressed in linear form using the probability chains technique described in (O'Hanley et al., 2013a).

4.2.2 Expected Accessible Upstream Habitat

Expected accessible habitat $\tilde{v}_j$ upstream of each barrier j was estimated as the sum of the expected accessible length of all river segments belonging to the upstream subnetwork of j. A subnetwork is defined as the area of river upstream of a barrier up to the next set of barriers or the river terminus. Hidden barriers are assumed to be uniformly distributed throughout the river network, so the probability that a hidden barrier is present along a specific river segment s is given by the ratio of the length ℓ_s of segment s, to total habitat length L ($\frac{\ell_s}{L}$). Assuming that there are n hidden barriers situated across the whole river network, the probability π_{skt} that k hidden barriers are located in river segment s, t hidden barriers are located downstream distance s (i.e., between s and the river mouth) and the $n-k-t$ remaining hidden barriers are located elsewhere in the river network is described by a multinomial distribution with counts k, t and $n-k-t$ and event probabilities $\frac{\ell_s}{L}$, $\frac{\ell_s^{'}}{L}$ and $\frac{\ell^{''}}{L}$, where $\ell_s^{'}$ is the length of river downstream of segment s and $\ell_s^{''}$ is the total length of river not directly downstream or within segment s. Expected accessible length $\tilde{\ell}_s$ of river segment s, in turn, can be calculated by combining probabilities $\tilde{\pi}_{skt}$ together with conditional expectations for the cumulative passability downstream of segment s and the effective length of segment s.

More precisely, let S, indexed by s, be the set of barrier-free, confluence bounded sections of river, each of uniform habitat type and quality. For each segment $s \in S$, we define:

$\ell_s =$ length of river segment s

$\ell_s^{'} =$ total length of river directly downstream from segment s

$\ell_s^{''} =$ total length of river not directly downstream of or within segment s (i.e., $L - \ell_s^{'} - \ell_s^{''}$)

The total length of river in the river network is denoted by $L = \sum_{s \epsilon S} \ell_s$. By assumption, $\ell_s > 0$ for all $s \in S$. In addition, let $\tilde{\ell}_s$ be the expected effective length of segment s taking into account the presence of hidden barriers. The probability that k hidden barriers are located in segment s and t hidden barriers are located downstream, with the remaining $n-k-t$ hidden barriers located elsewhere in the river network, is given by π_{skt}. The conditional expected cumulative passability of segment s given t hidden barriers are located downstream is denoted by $E(P_s|t)$, while $E(L_s|k)$ represents the conditional expected effective length of segment s given k hidden barriers are uniformly distributed along the length of segment s (i.e., P_s and L_s are both random variables representing, respectively, the cumulative passability and effective length of segment s). Each hidden barrier is assumed to have a mean passability of $\tilde{p}$.

Assuming the river network is composed of at least 2 segments ($|S| \geq 2$), then:

$$\tilde{\ell}_s = \sum_{k=0}^{n} \sum_{t=0}^{n-k} \pi_{skt} \cdot E(P_s|t) \cdot E(L_s|k) \tag{4.6}$$

where:

$$\pi_{skt} = \frac{n!}{k!t!(n-k-t)!} \left(\frac{\ell_s}{L}\right)^k \left(\frac{\ell_s^{'}}{L}\right)^t \left(\frac{\ell_s^{''}}{L}\right)^{(n-k-t)} \tag{4.7}$$

$$E(P_s|t) = \tilde{p}^t \tag{4.8}$$

$$E(L_s|k) = \sum_{r=0}^{k} \tilde{p}^r \left(\frac{\ell_s}{k+1}\right) \tag{4.9}$$

We note that, if $\ell_s^{'} = 0$ or $\ell_s^{''} = 0$, which implies $t = 0$ and $t = n-k$, respectively, then π_{skt} reduces to:

$$\pi_{skt} = \frac{n!}{k!(n-k)!} \left(\frac{\ell_s}{L}\right)^k \left(1 - \frac{\ell_s}{L}\right)^{(n-k)} \tag{4.10}$$

For all other values of t, $\pi_{skt} = 0$ whenever $\ell_s^{'} = 0$ or $\ell_s^{''} = 0$.

The expected accessible habitat, $\tilde{v}_j$, upstream of each barrier was calculated as the sum of the expected accessible length of all river segments belonging to U_j, the upstream subnetwork of barrier j:

$$\tilde{v}_j = \sum_{u \epsilon U_j} \tilde{\ell}_u \tag{4.11}$$

BAT toolbox for ArcGIS 10.2.1 was used to determine which segments compose networks U_j.

4.2.3 Data

Data on 6,989 natural and artificial fish passage barriers across Maine were obtained from the US Fish and Wildlife Service Gulf of Maine Coastal Program. Each barrier in this database is georeferenced, and

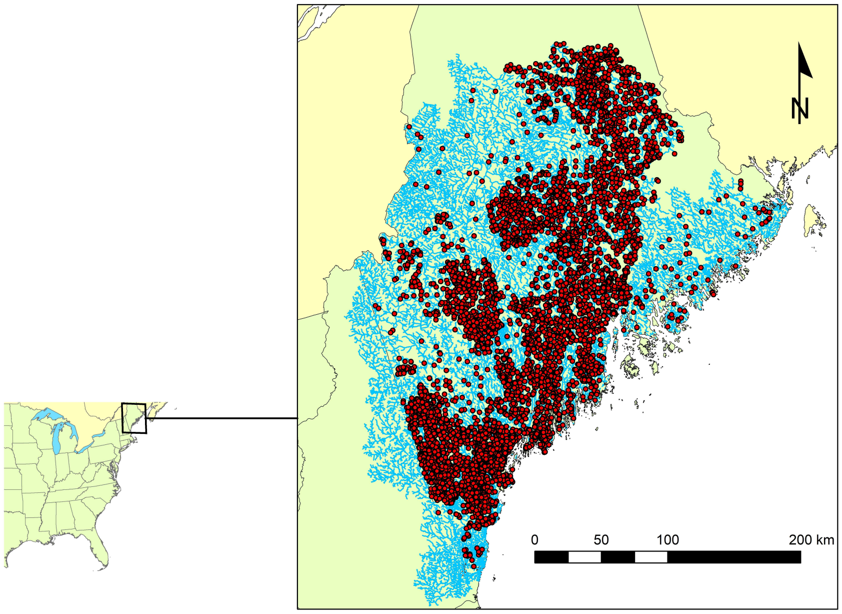


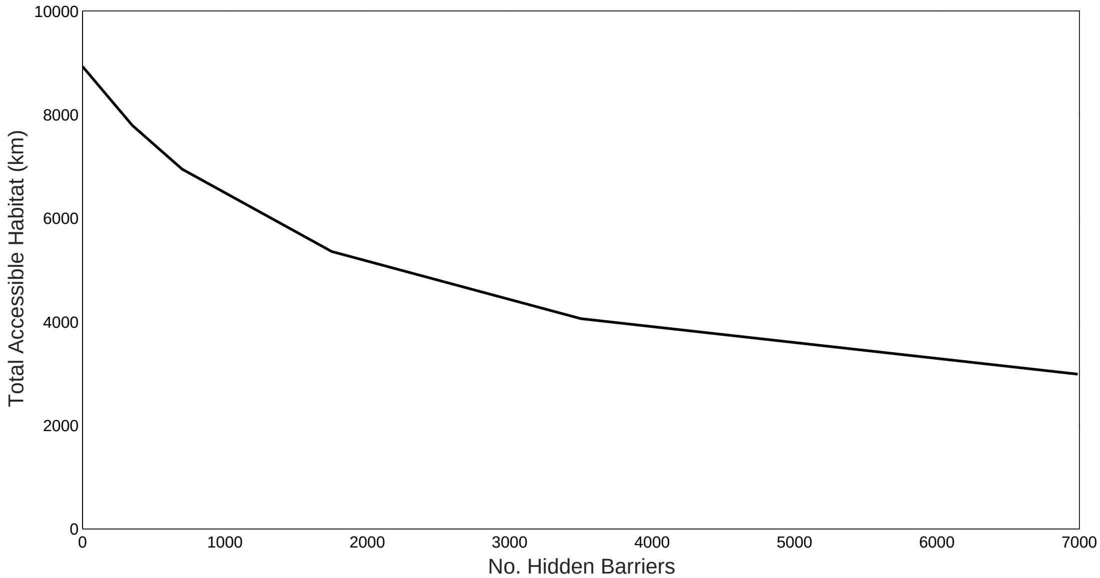
Figure 4.1: Barriers, both artificial and natural, in the state of Maine.

includes a description of its barrier type (dam, culvert) and a qualitative assessment of current passability (full or partial barrier). Passability values of 0 were assigned to full barriers and 0.5 to partial barriers. We considered only one mitigation option for each barrier which, if implemented, would increase the barrier passability either to 0.75 (for the large dams >25ft) or to 1 (for the small dams ≤25ft and culverts). The location of the 6,989 known barriers is shown in Figure 4.1.

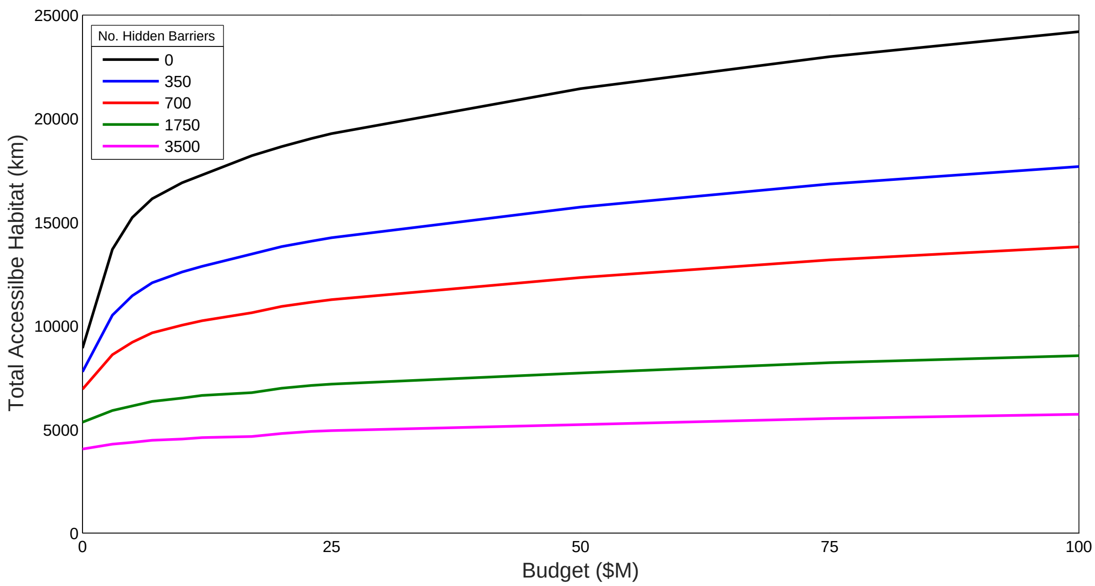
We assumed that the passability of each hidden barrier was equal to the median passability (0.5) of known barriers.

4.3 Results

In Maine watersheds, known natural and artificial barriers (Fig. 4.1) which block free movement of fish, allow access to just 18% of the 49,840 km of river length. But this accessibility level assumes that there are no hidden barriers anywhere in the watersheds. Taking into account the presence of hidden barriers dramatically affects the accessible habitat. Assuming, for example, the presence of 700 hidden barriers, which is approximately 10% of the number of known barriers, would cause accessible habitat to drop by almost a quarter (Fig. 4.2a). With 1,750 hidden barriers, (25% increase), current accessible habitat would decrease by 40%.



(a)



(b)

Figure 4.2: Current accessible habitat for increasing number of hidden barriers (a) and anticipated versus actual accessible habitat for an uninformed prioritization approach given different numbers of hidden barriers (b).

Our analysis shows that the presence of hidden barriers leads to huge shortfalls in anticipated gains for a prioritization method that ignores hidden barrier uncertainty. The actual total accessible habitat after the

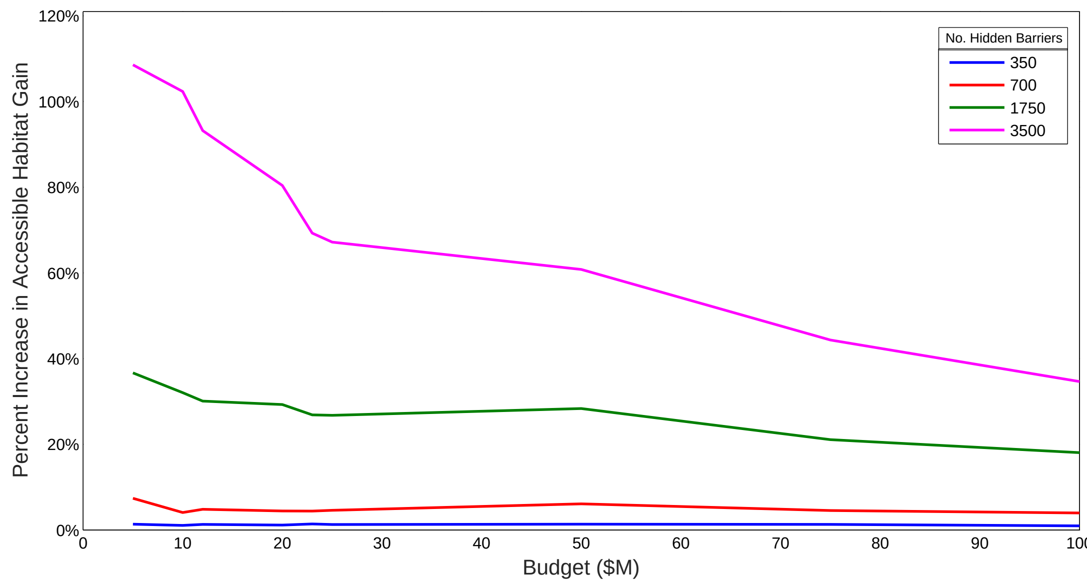
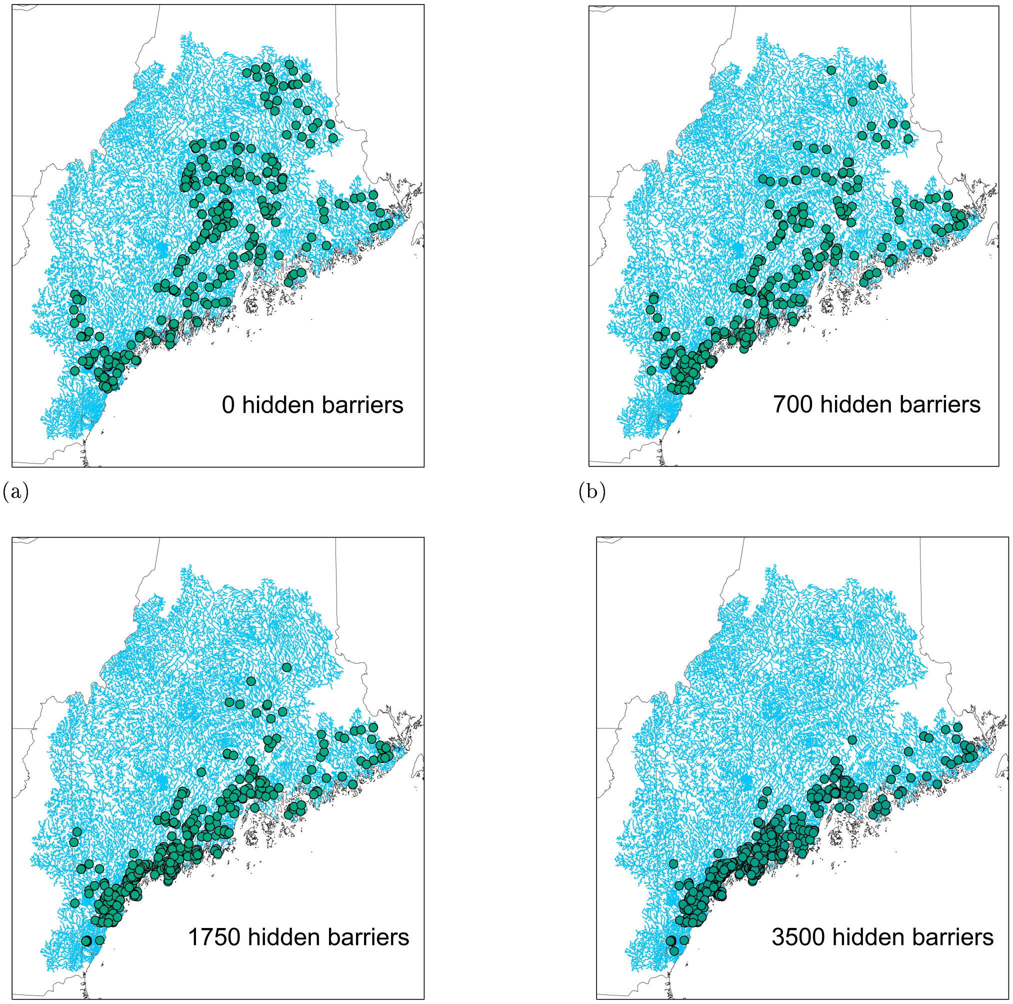


Figure 4.3: Percent increase in accessible habitat gain from factoring in hidden barrier uncertainty for various budget levels and different numbers of hidden barriers.

implementation of the selected mitigation actions, even when there is a small number of hidden barriers present and regardless of the available budget, drops dramatically (Fig. 4.2b). Assuming, for example, a 10% increase in the number of known barriers would mean that the anticipated accessible habitat would actually be 40% less than expected. With an increase of 50% the actual accessible habitat would be just a quarter of the anticipated one. Not accounting for hidden barrier uncertainty results to inaccurate expectations for the outcomes of mitigation strategies.

We find that anticipating for hidden barriers in the prioritization process can lead to significant improvement in accessible habitat gains. Our model, that optimizes barrier removal prioritization decisions while considering the effects of hidden barriers on the effective accessible habitat length and on the combined barrier passability, can increase the gains in reconnected accessible habitat in excess of 110%, when the available budget is low and the number of hidden barriers relatively large, compared to a conventional, uninformed approach (Fig. 4.3). When assuming a 50% increase in the number of barriers (3500 hidden barriers) the average increase in habitat gains across a range of budgets reaches 80%. With the presence of 1750 hidden barriers (25% increase) our model improves the potential gains in accessible habitat by 30% on average, highlighting the fact that even for a moderate increase in the number of known barriers our approach can boost the efficiency of the restoration planning.

Fig. 4.4 shows the spatial arrangement of solutions for $25M available budget and for 4 different hidden barrier scenarios. The spatial layout of the selected barriers for mitigation shows that the number of hidden



(a) (b) (c) (d)

Figure 4.4: Locations of selected barriers given a budget of \$25M for different hidden barriers scenarios.

barriers highly affects the barrier selection. In particular, as the number of hidden barriers increases the selected barriers move towards the river mouth (Fig. 4.5). The optimal solution if no hidden barriers were present would include barriers with an average distance to mouth of 100km. This distance would drop to just one fifth if there were 25% additional hidden barriers. Barriers closer to the sea affect the overall habitat accessibility the most as each barrier's passability affects the combined passability of all its upstream sites. Here, the objective is to maximize the total accessible habitat length so as the river network gets more disrupted (i.e., the total number of barriers increases) it is expected for the optimization model to select barriers closer to the river mouth.

We also analyzed how well solutions optimized for a specified number of hidden barriers perform when the actual number of hidden barriers differs. This sensitivity analysis (Fig. 4.6) shows that erring too low (assuming that there are only a few hidden barriers present) or too high (assuming that there are many

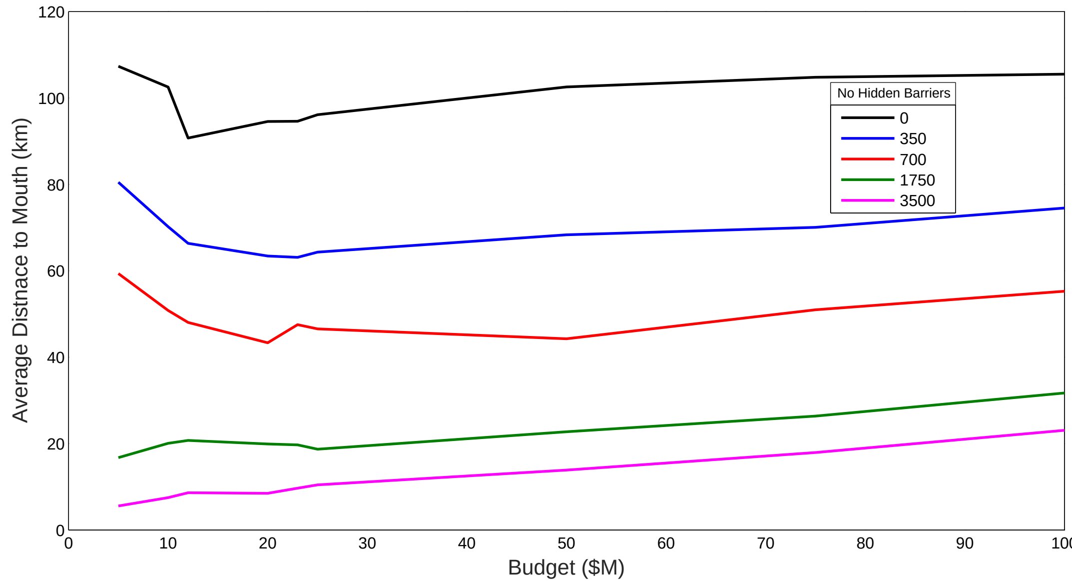
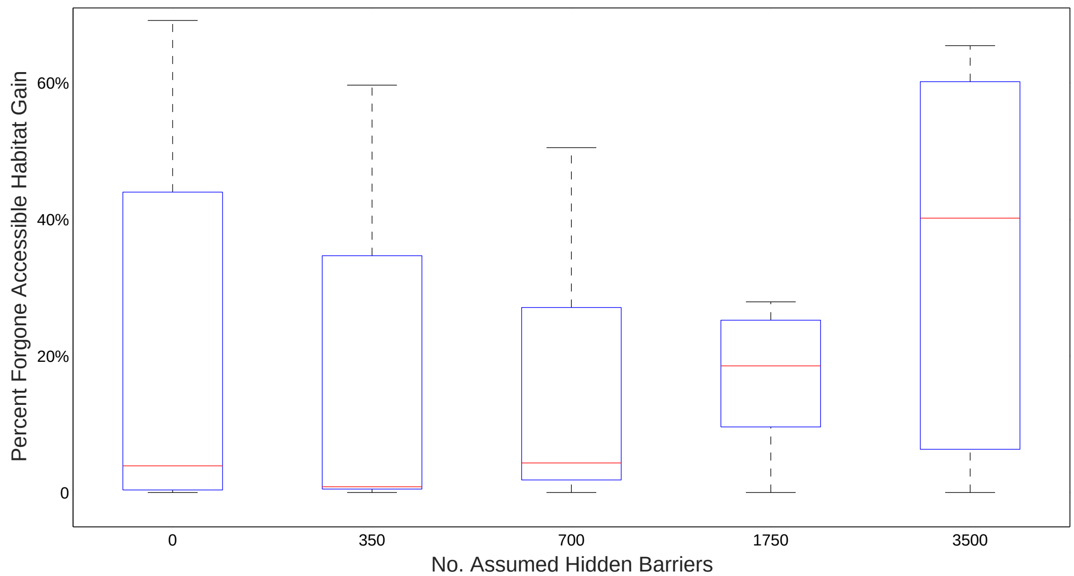


Figure 4.5: Average distance to mouth for various budget levels and different numbers of hidden barriers.

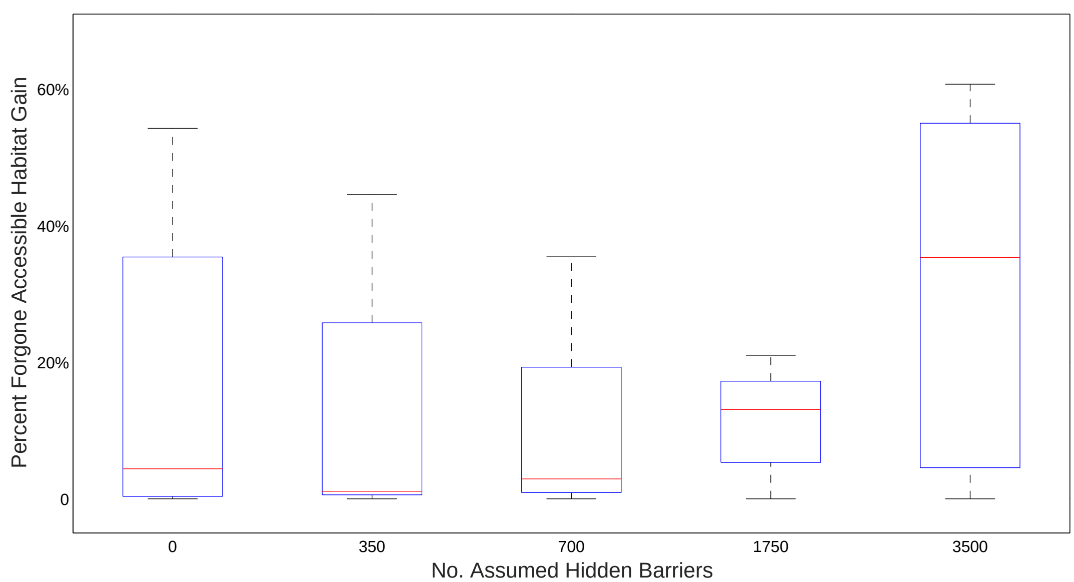
hidden barriers) leads to greater variability in terms of foregone habitat gains. In the worst case scenario the loss in accessible habitat gains for the two extreme cases (either no hidden barriers present or 50% additional barriers) can get as high as 69%. The most robust solutions are obtained when an intermediate number of hidden barriers is assumed, more specifically for solutions with 10% and 25% additional hidden barriers. The solution for 350 hidden barriers (5% increase) does the best on average but the solution for 1750 hidden barriers (25% increase) has the lowest variability. In particular, for 350 hidden barriers in most cases the percent of foregone habitat is less than 15%, however there are a few cases that it can get relatively high reaching 60% for $10M budget and 45% for $25M. For the 1750 hidden barriers the percent of foregone habitat gains varies less, not exceeding 28%.

4.4 Discussion

Our analysis shows that accounting for hidden barrier uncertainty is critical for maximizing the accessible habitat gains of barrier mitigation planning. As efforts to restore river connectivity are taking place in freshwater systems worldwide our findings can prove relevant to many restoration projects. Accounting for hidden barrier uncertainty gives a more realistic view of the potential outcomes of the barrier mitigation strategies, avoiding the huge shortfalls in anticipated gains that uninformed approaches face. Also, considering the existence of hidden barriers in the optimization process can substantially improve the actual gains in



(a)



(b)

Figure 4.6: Box plots of the median, lower/upper quartiles and minimum/maximum (wiskers) amount of foregone habitat gain when the number of hidden barriers varies from what was planned for given budgets of \$10M (a) and \$25M (b).

accessible habitat. We find that even a small increase in the number of barriers highly affects the barrier selection with solutions moving towards the river mouth as the number of barriers goes up. Assuming an intermediate increase in the number of known barriers has the lowest variability in terms of foregone habitat gains. Our results highlight the necessity of accounting for hidden barrier uncertainty in the river restoration planning.

With regard to future research, our modeling approach could be extended, depending on the available data, to also consider the dispersal behavior of resident fish and other aquatic organisms (Cote et al., 2009; O'Hanley, 2011; O'Hanley et al., 2013b), instead of concentrating only on migratory fish populations. The incorporation of fish population dynamics (Paulsen and Wernstedt, 1995; Zheng et al., 2009; Ziv et al., 2012) would also significantly enhance the sophistication and practicality of our approach.

River infrastructure serves many of society's needs, e.g., transportation, flood control, power production (Doyle and Havlick, 2009) so inevitably river restoration planning in reality involves many goals and constraints, often competing (Kemp and O'Hanley, 2010). An interesting extension of our approach could consider multiple objectives, like for example dam safety (Zheng and Hobbs, 2013), water storage and hydropower production (Kuby et al., 2005), potential threats from invasive species (Zheng et al., 2009). A multi-objective optimization approach that takes into account hidden barrier uncertainty could prove a very valuable tool for the river managers in the decision making process.

Another interesting line of research would be to include in our modeling approach a statistical analysis to predict the number of hidden barriers. Ramos (1999) suggests the use of Bayesian statistics to simulate the unrecorded number of events while Jeuland et al. (1980) and Fader and Hardie (2000) propose the use of beta-binomial/negative binomial distribution to model underreported count data. Predictions regarding the actual number of hidden barriers would greatly improve the effectiveness of barrier prioritization decisions.

Finally, an interesting possible extention of our framework would be to include a sensitivity analysis on assumed passability values of hedden barriers. We are currently using the median value of the passabilities of the known barriers. It would be interesting to explore how variations of this value would affect the optimal barrier mitigation strategies.

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Chapter 5

Conclusions

5.1 Summary of key contributions

The importance of river connectivity and the severe consequences that habitat loss and fragmentation have on the aquatic ecosystems are well established (Lucas and Baras, 2001; Fahrig, 2003; O'Hanley and Tomberlin, 2005). Restoring river continuity is considered essential for biological conservation and for improving the status of freshwater ecosystems (Roni et al., 2002; Fischer and Lindenmayer, 2007; Kemp and O'Hanley, 2010). River systems worldwide are heavily impacted (Bednarek, 2001) by the presence of large numbers of artificial barriers, such as small weirs, road crossings, culverts, sluices, tide gates and large hydropower dams (Kemp and O'Hanley, 2010). Decisions about installing new infrastructure (e.g., hydropower dams) or mitigating existing structures that block fish passage are complex and inevitably raise conflict between the need for healthier river systems and society's demand for ecosystem goods and services (e.g., for energy, transportation and flood control). River management needs appropriate tools in order to balance the competing goals and evaluate potential alternatives. Effective barrier placement and/or mitigation planning is critical for maintaining the integrity of freshwater ecosystems.

This thesis contributes to the literature by addressing key issues relevant to barrier placement and removal decisions that have not been previously examined. The findings of this thesis are expected to prove useful not only to researchers but also to practitioners involved in water policy and river management. The key contributions of the three papers presented in this thesis are discussed in more details below.

The first paper introduces a novel framework for optimally locating small hydropower plants (SHP). To date relevant studies focused almost exclusively on identifying feasible sites for SHP installation rather than optimizing location decisions. The proposed models maximize total hydropower production while limiting negative impacts on river connectivity. Importantly, the backwater effects that the installation of new SHP have on water surface profiles upstream are taken into account. In particular, the proposed models capture the interactive effects that SHP installation has on both power potential of nearby upstream sites and the ability of fish to successfully pass such sites. Our framework provides new insights on how hydrological issues can be incorporated in SHP location modeling, resulting in a more realistic approach to managing complex river systems. According to our findings installing new SHP fitted with fish passes in river networks already heavily fragmented can actually create a win-win situation where increasing hydropower generation also improves river connectivity.

The second paper presents an optimisation framework to prioritize barrier mitigation decisions for improving the viability of migratory fish populations. In the literature, optimisation studies that take into account fish dispersal and population dynamics as part of river restoration planning are often overly simplistic or very complex, but non-scalable. The framework presented in this thesis combines spatially explicit population viability analysis (PVA) with optimization techniques to prioritize barrier repair and removal decisions. Fish homing fidelity, straying behavior, and environmental variability (on population growth) are included in the modeling framework to assess the relative importance of incorporating spatiotemporal fish population dynamics into river connectivity restoration planning. Our analysis shows that the type of homing behavior has a significant effect on barrier mitigation decisions. In particular, with reach homing, almost the same sets of barriers selected for mitigation maximize population sizes and accessible habitat. With river homing, however, the barrier selection differs significantly between the two models. With this homing pattern there is no need to remove all barriers to maximize equilibrium population size. A stochastic version of our model reveals that removing all barriers actually results in a marginal increase in quasi-extinction risk.

The third paper deals with uncertainty related to the existence of unknown or "hidden" barriers when optimising river connectivity restoration actions. Barrier inventories are incomplete and this fact has been ignored in all relevant studies thus far. The novel optimisation framework introduced in this thesis prioritizes barrier mitigation decisions while accounting for the effects that "hidden" barriers have on the effective accessible habitat length and on the cumulative passability. We find that ignoring hidden barriers leads to huge shortfalls in anticipated accessible habitat gains. Also, according to our findings, anticipating for hidden barriers in the prioritization process can lead to significant improvement in accessible habitat gains. Finally, we find that solutions optimized for an intermediate number of unknown barriers perform well regardless of the actual number of hidden barriers. Accounting for hidden barrier uncertainty gives a more realistic view of the potential outcomes of the barrier mitigation strategies resulting in far more effective river restoration planning.

Optimization techniques are ideally suited for dealing with the multiple, often conflicting, environmental and socioeconomic goals involved in river restoration planning. The new insights into optimising barrier removal and placement decisions introduced in this study are expected to be valuable to both academics and practitioners, by way of improving the effectiveness and cost-efficiency of river restoration and development planning.

5.2 Future directions

With regard to future research there are several ways that our models could be improved or extended. For example, the SHP and the hidden barriers models, are concentrated only the dispersal needs of migratory fish populations where fish travel between fresh water and the sea. This is not the only type of migratory behavior. Our modeling framework could be adapted to handle potadromous dispersal patterns (Cote et al., 2009; O'Hanley et al., 2013) where fish move regularly between different sections of the river. The incorporation of fish population dynamics (Paulsen and Wernstedt, 1995; Ziv et al., 2012) would also significantly enhance the sophistication and practicality of the models.

Some possible extensions of the SHP models are: first, the SHP models could focus on installation decisions of larger, reservoir type hydropower dams instead of focusing on locating smaller run-of-river type hydropower dams. As the name implies, such dams create large reservoirs upstream (e.g., Lake Meade behind Hoover dam). Their main benefit is the much greater hydropower that can be generated. On the other hand, their impacts go well beyond disrupting river connectivity; they can significantly reduce sediment flow, dampen seasonal flow variation (aka the "natural hydrograph"), cause loss of riparian and terrestrial habitat (due to submersion), and promote the spread of aquatic invasive species (Stanford et al., 1996). At the same time, large reservoir dams can deliver additional socio-economic benefits that run-of-river dams at best only partially provide, such as water storage/supply, flood protection, fishing, and recreational opportunities (Kuby et al., 2005; Zheng et al., 2009). Both the socio-economic benefits and environmental costs of dams can be estimated fairly easily using established market and non-market valuation techniques (MacDonald et al., 2011), suggesting that one might consider integrating adopting a bio-economic analysis framework to optimize large hydropower dam location decisions.

Second, one could take a more integrated approach that considers hydropower dam placement together with artificial barrier mitigation decisions. Such a model would allow for offsetting actions in which reduced passability due the installation of hydropower facilities may be compensated for by improvements in passability at other locations (Owen and Apse, 2015). With such a framework, it would be possible to determine where best to carry out barrier mitigation, namely at newly installed hydropower dams or at other existing structures that more heavily impact connectivity. These sorts of considerations are important in many heavily developed river systems, such as the US, Canada, and Europe where conflict often arrises between proponents on each side of the renewable energy generation versus river connectivity restoration debate.

Finally, the SHP framework could be enhanced by including a sort of sensitivity analysis on the effects of the assumed fish passage efficiency on the installation decisions. In our case study we have assumed that SHPs would be fitted with fish passes having a combined upstream/downstream passage efficiency of 0.5, as it was suggested in Noonan et al. (2012). It would be interesting to explore how would the optimal installation scenarios would vary when fish passes would allow a different passability rate.

An interesting line for research for the proposed optimisation framework that explores how optimal barrier mitigation strategies are affected by the consideration of fish dispersal and population dynamics would be to consider a stage-structured population model and examine how habitat needs and dispersal dynamics at each life stage interact with barrier mitigation decisions. This modelling framework could also be adapted to consider additionally habitat quality, and how it is affected by pollution caused by the urban and industrial development. Our current framework considers only the accessible length of spawning habitat. It would be interesting to consider apart from habitat quantity its quality as well and explore how it would affect barrier prioritisation strategies.

Finally, the hidden barriers approach could be extended by: first, considering multiple objectives, like dam safety, water storage, hydropower production, potential threats from invasive species (Kuby et al., 2005; Zheng et al., 2009; Zheng and Hobbs, 2013). River infrastructure serves many of society's needs, e.g., transportation, flood control, power production (Doyle and Havlick, 2009) so river restoration planning inevitably involves many goals and constraints, often conflicting (Kemp and O'Hanley, 2010).

Second, the hidden barriers model could be extended by including in the modeling approach a statistical analysis to predict the number of hidden barriers. Ramos (1999) suggests the use of Bayesian statistics to simulate the unrecorded number of events while Jeuland et al. (1980) and Fader and Hardie (2000) propose the use of beta-binomial/negative binomial distribution to model underreported count data. This statistical analysis would improve the effectiveness of the barrier mitigation strategies.

A third possible extension of this framework could be to conduct a sensitivity analysis on the assumed passability values of the hidden barriers. We are currently using the median value of the passabilities of the known barriers. It would be interesting to explore how variations of this value would affect the optimal barrier mitigation strategies.

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