How to Choose? A Bioeconomic Model for Optimizing River Barrier Mitigation Actions

Steven King[a] Jesse R. O'Hanley[bc] Iain Fraser[cd*]*

[a] *UNEP World Conservation Monitoring Centre, UK*

[b] *Kent Business School, University of Kent, UK*

[c] *Durrell Institute of Conservation and Ecology, University of Kent, UK*

[d] *School of Economics, University of Kent, UK*

Abstract

River infrastructure can cause adverse impacts on fish populations, which, in turn, compromises the ability of river ecosystems to provide a range of ecosystem services. In this paper, we present a methodological approach to assess the potential economics costs and benefits of river connectivity enhancement achieved through removal and mitigation of fish dispersal barriers. Our approach combines the results of a stated preference study for nonuse values of rivers and statistical models of fish population responses to barrier mitigation actions within an integrated bioeconomic optimization framework. We demonstrate the utility of our methodology using a case study of the River Wey catchment in southeast England, which contains over 650 artificial barriers. Our results reveal the presence of benefit-cost trade-offs which can form the basis for river barrier mitigation policy development. In particular, we find that benefits exceed costs in the River Wey for all levels of investment in barrier mitigation considered (£2.5 to 53.4M). Furthermore, from an economic efficiency standpoint, a total budget of approximately £22.5M allocated to barrier mitigation would maximize net societal benefits derived from anticipated increases fish species richness and abundance.

Keywords: fish passage barriers, river connectivity, discrete choice experiments, bioeconomic modeling, optimization, cost benefit analysis.

1 Introduction

River systems deliver a range of ecosystem service benefits for human and economic activity (Doherty et al., 2014; Gopal, 2016). Many of these services are provided by healthy fish populations (Holmlund and Hammer, 1999). However, more than 50% of rivers globally are impacted by physical infrastructure (e.g., dams, weirs, and culverts) that disrupt the longitudinal connectivity and obstruct fish from accessing essential habitats and resources. Numerous studies have demonstrated the negative effect that artificial barriers have on migratory (Catalano et al., 2007; Lucas et al., 2009; Gough et al., 2018; O'Hanley et al., 2020) and resident fish

*Correspondence email: `i.m.fraser@kent.ac.uk`

populations (Nislow et al., 2011), including restricted range, altered population structure, reduced spawning and recruitment success, genetic isolation, and local extinction (Wofford et al., 2005; Nunn and Cowx, 2012; Barbarossa et al., 2020; Pereira et al., 2020). This, in turn, can compromise the ability of river ecosystems to deliver the full range of ecosystem services (Rounsevell et al., 2018). Despite the importance of healthy inland fish populations for delivering river ecosystem benefits, it is conspicuous that they are not included as part of the United Nations Sustainable Development Goals (Lynch et al., 2020).

Improving river connectivity through removal, repair, or modification of fish passage barriers has been demonstrated to deliver increased fish density (Gardner et al., 2013; Birnie-Gauvin et al., 2017, 2020), diversity (Catalano et al., 2007), and rapid colonization of formerly inaccessible stream reaches (Roni et al., 2008). Accordingly, a number of legislative drivers for mitigating the impacts of these barriers now exist like the EU Water Framework Directive (WFD) (Kemp et al., 2008) and the US Endangered Species Act (Pohl, 2002). The recent EU Biodiversity Strategy, for example, explicitly recognizes the need to remove fish passage barriers by committing to restore at least 25,000km of free-flowing rivers by 2030 (EC, 2020).[1] Traditionally, environmental legislation has been predicated on the basis of scientific evidence and ethical values (Turner and Daily, 2008). However, it is now widely acknowledged that the range of economic services that ecological systems provide, including river ecosystems, contribute significantly to human welfare and should form a material consideration in policy making (Gopal, 2016). Thus, given the legislative requirements to protect the environment, environmental agencies have sought methodologies that can effectively and efficiently maximize ecological returns given associated costs and benefits. In response, various studies have been undertaken to devise more effective policy responses (MEA, 2005; TEEB, 2008; NEA, 2011; Rounsevell et al., 2018). A key feature of these studies has been the development of frameworks in which economic valuation of ecosystem services is undertaken so as to identify specific ecosystem services contributing to human well-being (Bateman et al., 2011). In principal, what this means is that information about ecosystem services needs to be collected and analyzed so that cost benefits analysis (CBA) of policy options can be carried out by government agencies when formulating and administering environmental policy (Johnston and Rosenberger, 2010). For instance, the WFD specifically requires CBA in catchment management plans in order to direct an efficient allocation of resources for environmental protection (Hanley et al., 2006). However, as noted by Logar et al. (2019) actual examples of CBA for river restoration projects, such as barrier removal, are limited. The lack of actual CBA studies is not due to any lack of benefit estimates for river restoration. Instead, Logar et al. (2019), who cite only a handful of existing CBA studies published in the literature, argue it is the lack of cost data for river restoration that is the limiting factor. We add to this literature by focusing on a study site for which economic cost and benefit information as well as detailed fish population data are available.

Specifically, in this paper, we investigate how to use ecosystem service information about a river to efficiently target barrier mitigation actions in order to optimize the delivery of river ecosystem services. To identify an efficient allocation of resources, we develop a bioeconomic model that simultaneously identifies how to maximize increases in fish species richness and abundance given available funds as well as estimate the economic benefits derived from improvements in these two biophysical attributes.[2] By subsequently combining costs and benefits, we are able to identify an efficient economic solution for barrier mitigation. Our framework involves integration of several related but independent research methodologies. First, we assess fish survey data

[1] We note recent evidence that the level of river fragmentation across Europe is significantly higher than previously recorded (Belletti et al., Under Review) suggests that this may be a considerable challenge requiring a significant amount of funding to achieve.

[2] We note that there can sometimes be unintended consequences from undertaking barrier mitigation as discussed by McLaughlin et al. (2013).

from our case study area, the River Wey, England, to understand current fish population status as well as what can potentially be achieved from barrier mitigation. From this, we estimate study area specific societal willingness-to-pay (WTP) for gains in fish species richness and species abundance using a stated preference discrete choice experiment (DCE). Second, we use fish survey data to develop and parameterize statistical models of predicted fish species richness and abundance responses to barrier mitigation. Third, we combine WTP estimates from our DCE with the fish population response models into a scalable mixed integer linear program (MILP) to optimize barrier mitigation decisions. Our integrated methodology combines research on the use of MILP in barrier mitigation planning with established ecological modeling and environmental stated preference valuation techniques. Our framework can readily facilitate CBA of specific river barrier mitigation scenarios at catchment scales.

This research presented here contributes to the existing literature on systematic approaches for river connectivity enhancement. Within this literature, there exists a growing number of examples employing optimization based approaches to maximize the amount of functional habitat available for fish (Erős et al., 2018; McManamay et al., 2019; O'Hanley et al., 2020). There are also papers that derive cost-effective solutions to optimize one or more fish population and socioeconomic metrics (King et al., 2017; Roy et al., 2018). A comprehensive review of how optimization has been applied in barrier mitigation planning is provided by McKay et al. (2017). However, our approach to integrating stated preference value estimates derived from a DCE combined with empirical fish population response models into an optimization framework makes several new contributions to the literature. First, we extend the approach developed by King et al. (2017) by examining the benefits as well as the costs associated with river barrier mitigation. There are a number of DCE studies undertaken that examine the potential societal benefits derived from river restoration activities (Doherty et al., 2014; Brouwer et al., 2016; Bergstrom and Loomis, 2017; Brouwer and Sheremet, 2017; Logar et al., 2019; Kunwar et al., 2020; Symmank et al., 2020) as well as barrier removal specifically (King et al., 2016). Importantly, our analysis only explicitly takes account of the benefits from barrier mitigation as valued by the population living in close proximity to the River Wey.[3]

Second, we consider changes in costs and benefits from barrier mitigation by simultaneously examining two environmental outcomes – changes in species richness and species abundance. This means that we are considering changes in multiple environmental indicators and, as such, need to ensure that our stated preference WTP estimates are aligned with this approach. As is clear from much of the existing stated preference literature on river restoration, estimates of WTP are frequently predicated on somewhat vague definitions of environmental improvement. It is also the case that the valuation literature tends to place more emphasis on use values than non-use values (Logar et al., 2019; Kunwar et al., 2020) which in part explains the findings reported by Bergstrom and Loomis (2017) that annual WTP is positively related to geographical scale of the restoration activity. Third, our integrated modeling approach provides a template for improving the evaluation and implementation of water related policies such as the WFD. The need for such approaches can be motivated by existing criticism of the WFD in terms of the limited use of economic analysis (Berbel and Expósito, 2018). These criticism in part reflect the difficulties inherent in integrating economic information into water management. It is also the case that achieving integrated water management via the WFD requires a considerable degree of coordination, as noted by Junier and Mostert (2012). The integrated modeling framework we present here can be viewed as a useful tool in helping to improve coordination

[3]Another approach to valuing the benefits of river restoration is presented by Baggio et al. (2020). By examining actual fish catch data from anglers before and after river restoration activities, they were able to estimate the increase in value derived by anglers from a change in river management.

of policy implementation. It is also the case that our integrated approach to examining barrier removal provides a sound platform upon which to undertake interdisciplinary analysis of river restoration, which is also identified in the literature as a necessary requirement for achieving better policy outcomes (Grabowski et al., 2018).

Overall, our results indicate that the benefits of barrier removal within the River Wey significantly outweigh the associated costs. Indeed, we are able to identify that it would be economically efficient to invest £22.5 million in barrier mitigation actions. At this funding level, marginal costs equal marginal benefits. This finding is in keeping with those reported by Logar et al. (2019) who report that benefit cost ratios for river restoration are frequently high. However, unlike the results reported by Logar et al. (2019), our benefit estimates are not confined to *in situ* use activities by local communities. In generating this result, not only are we able to demonstrate the economic case for barrier removal but we are also able to show the potential usefulness of our integrated modeling approach.

The remainder of the paper is organized as follows. In Section 2, we begin by briefly describing our case study area and the DCE used to derive WTP estimates. This is then followed by an explanation our fish population response models and how model parameters were obtained. This subsequently followed by a detailed description of our MILP bioeconomic model. Finally, in Section 2, we provide an overview of barrier mitigation cost estimates used in the bioeconomic model. In Section 3, we present and discuss statistical results for the DCE and fish population response models as well as CBA of barrier mitigation in the River Wey. In Section 4, we provide some concluding remarks and observations.

2 Methods

2.1 Case Study Area

The River Wey catchment, located in the southeast of England, covers an area of 904km^2. The Wey is comprised of two main branches that join near Guildford before eventually flowing into the non-tidal portion of the River Thames close to Weybridge (see Figure 1 for details). Agriculture is the principal land-use in the south and west of the catchment, while the north part of the catchment is primarily urban (EA, 2008a). Recreational fishing is widespread throughout the Wey catchment. As is typical in England, numerous angling clubs (more than 30 in the Wey) hold private fishing rights to a majority of accessible reaches. Besides fishing, recreational boating is another popular activity, with approximately 34 miles of historic canals located in the middle and lower portions of the river system.

The Wey catchment includes a variety of habitats (e.g., heathland, woodland, and watermeadow) that support a diversity of aquatic and terrestrial wildlife. Sections of the river are protected as Sites of Special Scientific Interest (SSSI) or as part of nature reserves. Based on fish surveying work by the Environment Agency (EA), a total of 19 native fish taxa are present in the River Wey, including species valued by anglers (e.g., common carp, pike, bream, and perch) and a number of species of conservation concern (e.g., brown/searun trout, barbel, and European eel).

One of the main threats to fish and othe aquatic species in the River Wey is the presence of artificial river barriers, which negatively impact river connectivity and ecosystem function. Over 800 river barriers, including dams, weirs, sluices, canal locks, culverts, and natural waterfalls, are present across the catchment

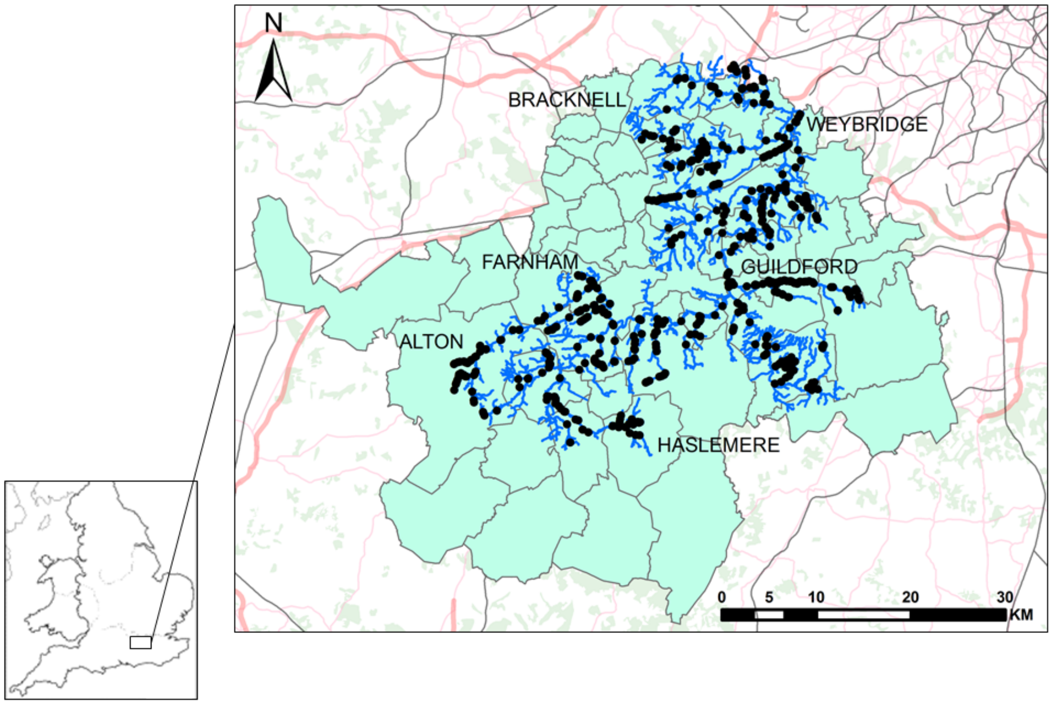


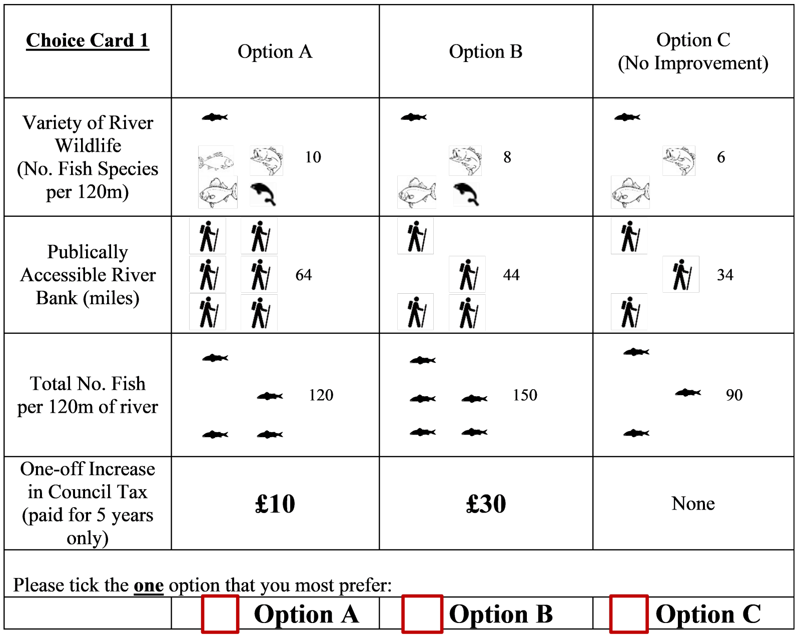
Figure 1: Location and extent of River Wey catchment. Artificial barriers are represented by small black dots. Green shaded areas represent postcode boundaries of the benefiting population.

(King et al., 2017). As part of its action plan for improving the ecological potential of the River Wey, the EA, the main public body responsible for managing water bodies in England and Wales, has identified the mitigation of fish passage barriers as a top priority(EA, 2008b). Barrier mitigation refers to any number of options designed to increase the "passability" of barriers, with passability typically measured as the proportion of fish able to successfully pass a barrier in the upstream and or downstream direction (Kemp and O'Hanley, 2010). Common types of mitigation include retrofitting or replacing stream crossings and installation of fish passes (*aka* fish ladders), modification (e.g., notching), or partial/full removal of dams, weirs, and other similar structures.[4]

2.2 Non-market Benefits of Barrier Mitigation

In this paper, we employ estimates of WTP for local river ecosystem improvements derived from barrier mitigation actions using a the results of DCE presented by King et al. (2016). In brief, fish species richness (Var_Wild) and fish abundance (Tot_Fish) were selected as the two biophysical river quality attributes for inclusion in the DCE. This choice was based on reviewing numerous studies that show significant and often rapid increases in fish species and abundance follow barrier mitigation actions (Catalano et al., 2007; Burroughs et al., 2010). Critically, fish species richness and abundance can also be directly linked with various ecosystem goods and services, including recreational fishing and tourism, iconic species viewing, the existence value of native wildlife, educational opportunities, and mental/physical health.

[4] We note that there are options to improve river connectivity that do not always require the removal of dams such as various forms of fish passage. However, as Kemp (2016) explains, the potential benefits they offer for fish passages are far from clear as they do not offer an effective catch all mitigation strategy.



Choice Card 1	Option A	Option B	Option C (No Improvement)
Variety of River Wildlife (No. Fish Species per 120m)	10	8	6
Publically Accessible River Bank (miles)	64	44	34
Total No. Fish per 120m of river	120	150	90
One-off Increase in Council Tax (paid for 5 years only)	**£10**	**£30**	None
Please tick the **one** option that you most prefer:			
	Option A	**Option B**	**Option C**

Figure 2: Example choice card.

In the introductory information that was provided as part of the DCE survey, respondents were provided background information about the River Wey and informed of a list of ecosystem goods and services that would improve as a result of increases in the fish species richness and abundance. Identified ecosystem goods were separated into two groups, one for fish richness, the other for fish abundance. King et al. (2016) took this approach so as to ensure that respondents' preferences for increased richness (e.g., the existence value of native wildlife) were not confounded with those for fish abundance (e.g., improved recreational fishing opportunities) and subsequently allowed us to isolate the welfare benefits for the two attributes separately. They also included public access to river bank (*Access*) as an additional attribute in the DCE to reduce informational or focusing biases and to capture respondents' preferences for footpaths next to the river. Finally, given the local nature of the study, a locally administered payment mechanism, namely a council tax increase collected yearly for a fixed 5-year period, was chosen to represent the cost attribute for the river barrier mitigation program (*Cost*).

The DCE is standard in that respondents were presented with a series of choice cards (see Figure 2) in which they were asked to choose between three options comprising two hypothetical river improvement options (options A and B) that provided an increase in at least one attribute at a cost and a status quo option (option C) with no attribute improvement and zero cost.

Attribute levels for fish species richness and abundance were based on EA fish survey data (see below). The richness attribute spanned the range for the observed number of fish species in a 120m stretch of river (6, 8, 10, 12). A proportionate scale was adopted for the fish abundance attribute (90, 120, 150, 180). The cost vector went from zero to the maximum cost for river barrier mitigation (i.e., the cost to mitigate all known barriers in the system) on a per capita basis per year for five years (£0, £5, £15, £30, £50). The access attribute was informed by currently available miles of riverside access and additional miles of access the cost vector could likely provide (34, 44, 54, 64). A main effects factorial design was generated for the DCE using standard software (Ngene 1.1.1, Choicemetrics, 2012). The final design comprised 24 different choice cards separated into four different choice blocks each containing six choice cards. Each respondent was presented with one of the blocks. The DCE was administered by a market research company to a panel of online respondents residing in postcodes located within approximately 10km of the River Wey (see Figure 1). In total 206 usable survey responses were obtained, yielding a total of 1236 (206×6) choice observations.

As is common King et al. (2016) employed a random utility model to obtain WTP estimates for increased fish richness and abundance. In keeping with the literature they assumed that the random utility model is specified in two parts: an observable deterministic component and an unobservable random component. It then follows that a respondent i makes a specific choice from a finite set, in this case options A, B and C. The utility (U) individual i obtains comes from selecting alternative j from choice set t as represented by equation (1):

$$U_{ijt} = \boldsymbol{\beta}_i' \mathbf{x}_{ijt} + \varepsilon_{ijt} \tag{1}$$

where $\mathbf{x}_{ijt}$ is a vector of attribute values, $\boldsymbol{\beta}_i$ is the vector of parameters (i.e., marginal utilities) for the set of attributes that are estimated, and ε_{ijt} is the unobservable random component assumed to be type one extreme value distributed (Train, 2009). For our particular application, King et al. (2016) specified the deterministic portion of the utility function as:

$$v_{ijt} = \beta_{1i} ASC_{ijt} + \beta_{2i} Var\_Wild_{ijt} + \beta_{3i} Access_{ijt} + \beta_{4i} Tot\_Fish_{ijt} + \beta_{5i} Cost_{ijt} \tag{2}$$

where ASC_{ijt} is an alternative specific constant for alternative j that takes the value 1 if the status quo (option C) is selected, 0 otherwise.

To recover the β parameters in equation (2), King et al. (2016) employed a random parameters logit (RPL) model, which has the advantage of considering the panel structure of the data and also allowing the β parameter estimates to vary across respondents so that individual preference heterogeneity can be captured. As an RPL model has no closed form, it estimated using simulation by repeatedly drawing values for the βs from prespecified distributions (in this case the normal distribution) and then maximizing the simulated likelihood function across the entire sample of respondents. Finally, we note that King et al. (2016) assumed that the cost attribute to be a fixed parameter, such that the resulting marginal WTP estimates are derived in the standard way as the ratio of the attribute estimates (i.e., β_2 for fish richness and β_4 for fish abundance) divided by the cost estimate coefficient (i.e., β_5).

2.3 Fish Richness and Abundance Responses to Barrier Mitigation

To estimate changes in fish species richness and abundance in response to barrier mitigation actions, statistical regression models were developed and parameterized using fish survey data collected by the EA. Our initial dataset consisted of total of 145 fish surveys completed at 44 sites across the Wey catchment from October 1989 to October 2011 using standard electrofishing methods. The mean length and area of each survey is approximately 120m and 1000m^2, respectively, with an average of approximately six species and 96 fish recorded per survey. In all, 19 different native species are present in the catchment (correction to King et al., 2017). After excluding older surveys conducted prior to 2002 and outliers with one or zero species recorded, our final data set consisted of 121 observations for species richness and abundance at 34 different survey sites.[5]

An underlying assumption of our regression models is that the effects of barrier mitigation on fish richness and abundance responses are mediated through increases in longitudinal river connectivity. A variety of metrics have been proposed for measuring longitudinal connectivity (Cote et al., 2009; Erős et al., 2011; McKay et al., 2013). For our purposes, however, we chose the C metric (Diebel et al., 2015), which accounts for the quality and accessibility of multiple river habitat types, as well as travel distances between each and every pair of habitat patches within a river network. In our implementation, a habitat patch corresponds to a river *subnetwork*, with a subnetwork defined as the section of river upstream of a barrier up to the next set of barriers or river terminus. As our river network has a strictly dendritic structure (i.e., never diverges in the downstream direction, thereby excluding multi-threaded river systems), there is a one-to-one correspondence between barriers and subnetworks with each river subnetwork uniquely identified by its immediate downstream barrier.

A key parameter of the C metric is d_0, which denotes the typical dispersal distance of the focal fish species/taxa/guild of concern, in our case, adult brown trout (*Salmo trutta*).[6] This parameter controls the distance decay function D_{jk} for any given pair of subnetworks j and k separated by a distance d_{jk}:

$$D_{jk} = \frac{1}{1 + \left(\frac{d_{jk}}{d_0}\right)^2} \tag{3}$$

The decay function is used to scale the relative amount of habitat provided by nearby subnetworks toward one and more more distant subnetworks toward zero.

With this in place, we employed equation (4), a log-linear model for predicting species richness as a function of subnetwork-level longitudinal river connectivity:

$$\ln(\bar{R}_\ell) = \alpha_0 + \alpha_1 C_\ell^{10\text{km}} + \alpha_2 \sqrt{USL_\ell} + \alpha_3 [RUNS \times AREA]_\ell \tag{4}$$

Here, $\bar{R}_\ell$ is the number of non-recorded or 'absent' species for a given survey ℓ (i.e., $\bar{R}_\ell = R^{\max} - R_\ell$, where R_ℓ is the recorded richness and $R^{\max}$ is total number of species in the study area), $C_\ell^{10\text{km}}$ is the current value of the C metric with a 10km dispersal distance (i.e., with $d_0 = 10$km) for the subnetwork in which a survey was conducted, USL_ℓ is the total length (in km) of river within or upstream of the subnetwork in which a survey was conducted, $[RUNS \times AREA]_\ell$ is the number of electrofishing runs ($RUNS$) performed during a survey round times the area ($AREA$) of a survey (in m^2), and the αs are the regression model parameters to be estimated. Note that while it is certainly possible to directly model species richness R_ℓ, we observed that better fitting models were obtained by using species absence $\bar{R}_\ell$ as the dependent variable.

As species absence $\bar{R}_\ell$ is necessarily a count variable, we employed Poisson regression to find maximum likelihood estimates for the α parameters in equation (4). More specifically, to avoid the overly restrictive assumption of equal mean and variance (aka equidispersion) imposed by a Poisson model, we used a generalization of the Poisson model (Consul and Jain, 1973) that incorporates a scaling factor θ to match observed

[5] The availability of fish survey data is important in our analysis as it allows us to establish a benchmark against which we can assess changes in fish abundance and richness as a result of barrier removals. It has, however, been noted in the literature (Bouleau and Pont, 2015) that the lack of clarity with regard to agreed reference conditions for key ecological measures within rivers has hindered the implementation of the WFD.

[6] We recognize that the specification of a focal species can have a strong influence on the barrier prioritization process. However, since our main concern is overall species richness the choice is largely arbitrary (i.e., selection of a different focal species will affect how connectivity is quantified, but will not qualitatively affect richness predictions).

variance. Values for $\theta = 1$ indicate equidispersion (variance equal to the mean), $\theta > 1$ overdispersion (variance is greater than the mean), and $\theta < 1$ underdispersed (variance is smaller than the mean).

For fish abundance, we used the following linear model to predict fish density as a function of subnetwork-level longitudinal river connectivity:

$$D_s = \gamma_0 + \gamma_1 C_s^{0.1\text{km}} [RUNS \times WIDTH]_s^2 \tag{5}$$

where D_s is mean fish density (per m) at a given survey site s, $C_s^{0.1\text{km}}$ is the current value of the C metric with a 100m (i.e., with $d_0 = 0.1$km) dispersal distance for the subnetwork in a survey site is located, $[RUNS \times WIDTH]_s$ is the mean of the number survey runs ($RUNS$) times the width ($WIDTH$) of a survey (in m) at a given site, and γ_0 and γ_1 are the regression model parameters to be estimated. Note that unlike the model for predicting species richness (4), which used all 121 survey observations, the model for species abundance (5) was estimated based on average abundance at the 34 survey sites. Given the nature of the dependent variable, we estimated this model employing standard ordinary least squares (OLS). Both statistical models (4) and (5) were fit using LIMDEP version 10 (Greene, 2012).

It is worth pointing out that in both models, the main independent variable of interest is longitudinal river connectivity, but measured at very different scales: 10km for richness and 100m for abundance. Intuitively, the different dispersal distances for the two models makes sense, with richness typically being more strongly influenced by broad-scale habitat access (10km dispersal distance) and abundance influenced more by local-scale habitat access (100m dispersal distance). In the case of richness, the model further includes corrections for the size and relative position of where surveys were conducted (USL_ℓ) and sampling intensity ($[RUNS \times AREA]_\ell$). In the case of abundance, only a correction for sampling intensity ($[RUNS \times WIDTH]_s$) is included.

2.4 Bioeconomic Model

To strategically target barrier mitigation actions in the River Wey, we integrate the economic and fish population modeling components described above into a bioeconomic optimization framework. Optimization, in the context of river restoration planning, aims at finding the most efficient allocation of limited resources to maximize restoration gains. It is particularly well-suited for dealing with problems involving a large number of interlinked decisions in a systematic and objective manner. For the current study, the bioeconomic optimization model was designed to select a portfolio of barrier mitigation actions that maximizes the economic benefits of increased fish species richness and fish abundance subject to a budget on the total cost of barrier mitigation. The logic behind the bioeconomic model is as follows. It is assumed that the river catchment of interest can be represented as a dendritic ecological network formed by a set of river subnetworks, each of which can be designated by its immediate downstream barrier (King and O'Hanley, 2016). Barriers selected for mitigation induce changes in the connectivity status of the river subnetworks (in our case measured using the C metric) due to increased habitat accessibility. Increased subnetwork connectivity, in turn, leads to increased fish richness and abundance (described in §2.3), the monetary benefits of which can be quantified by applying WTP estimates for increased fish species richness and abundance (described in §2.2) for a given level of investment in mitigation actions. From this, a Pareto efficient frontier can ultimately be constructed showing how societal benefits of barrier mitigation vary with cost and an optimal level of investment identified that maximizes total net benefit.

To develop a general mathematical formulation of our model, let J, indexed by j, be the set of fish passage barriers/subnetworks that are present within the planning area. Parameter h_j denotes the amount of habitat (measured in terms of length or area) in subnetwork j and parameter H the total amount of habitat in the river network (i.e., $H = \sum_{j \in J} h_j$). The set of mitigation projects available at barrier j (possibly empty) is given by S_j and indexed by i. The cost of implementing project i at barrier j is represented by c_{ji}, while the budget for carrying out mitigation is b. The number of households potentially benefiting from barrier mitigation is given by N^{houses} and the time horizon (in years) over which benefits accrue is given by N^{yrs}. The implicit prices that households are willing-to-pay for increased fish richness and abundance are denoted by WTP^{rich} and WTP^{abund}, respectively. Finally, the decision variables of the model are given by:

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

$C_j^d =$ the C metric connectivity of river subnetwork j evaluated at a dispersal distance of d

$R_j =$ mean fish species richness in river subnetwork j

$D_j =$ mean fish density in river subnetwork j

$B^{\text{rich}} =$ benefit of increased fish richness obtained from barrier mitigation

$B^{\text{abund}} =$ benefit of increased fish abundance obtained from barrier mitigation

$TB =$ total benefit obtained from barrier mitigation

With this in place, a nonlinear integer programming formulation of our optimization model is given below.

$$\max TB = B^{\text{rich}} + B^{\text{abund}} \tag{6}$$

$s.t.$

$$B^{\text{rich}} = N^{\text{yrs}} \times N^{\text{houses}} \times WTP^{\text{rich}} \times \frac{1}{H} \sum_{j \in J} h_j R_j \tag{7}$$

$$R_j = f(C_j^{d_1}, \boldsymbol{\pi}_j) \qquad \forall j \in J \tag{8}$$

$$B^{\text{abund}} = N^{\text{yrs}} \times N^{\text{houses}} \times WTP^{\text{abund}} \times 120 \times \frac{1}{H} \sum_{j \in J} h_j D_j \tag{9}$$

$$D_j = g(C_j^{d_2}, \boldsymbol{\mu}_j) \qquad \forall j \in J \tag{10}$$

$$C_j^d = F(j, d, \boldsymbol{x}) \qquad \forall j \in J, d \in \{d_1, d_2\} \tag{11}$$

$$\sum_{j \in J} \sum_{i \in S_j} c_{ji} x_{ji} \leq b \tag{12}$$

$$\sum_{i \in S_j} x_{ji} \leq 1 \qquad \forall j \in J \tag{13}$$

$$x_{ji} \in \{0, 1\} \qquad \forall j \in J, i \in S_j \tag{14}$$

The objective function (6) maximizes total economic benefits from barrier mitigation, which is the sum of benefits from increased fish richness B^{rich} and increased fish abundance B^{abund}. Equation (7) gives the expression for increased fish richness benefits B^{rich}, which is calculated as mean richness across all subnetworks in the river system $\frac{1}{H}\sum_{j\in J} h_j R_j$ multiplied by the WTP for increased richness WTP^{rich}, the number of benefiting households N^{houses}, and the time horizon N^{yrs}. Note that for our case study, we set N^{yrs} equal to 5 years to match the hypothetical length of the payment mechanism described in the DCE survey. Also note that while we do not discount benefits, this could be easily done by including an appropriate discount factor in the model.[7]

Equations (8) specify that mean species richness R_j in each subnetwork j is assumed to be some function $f(\cdot)$ of connectivity status $C_j^{d_1}$ evaluated at a dispersal distance d_1 and a vector of additional environmental covariates $\boldsymbol{\pi}_j$ influencing species richness. In our implementation, we set $f(C_j^{\text{10km}}, \boldsymbol{\pi}_j) = R^{\max} - \bar{R}_j$, with $R^{\max}$ being the total number of species in the study area and $\bar{R}_j$ being an estimate for the number of absent species in subnetwork j given by:

$$\bar{R}_j = \exp(\alpha_0' + \alpha_1 C_j^{\text{10km}}) \qquad \forall j \in J \tag{15}$$

The formula for $\bar{R}_j$ derives directly from equation (4) with $\alpha_0' = \alpha_0 + \alpha_2\sqrt{USL_j} + \alpha_3[\overline{RUNS \times AREA}]$ and $\overline{RUNS \times AREA}$ being the mean of survey runs times survey area across all surveys in our dataset. Although equations (15) are nonlinear, they can be easily approximated using a piecewise linear curve as described in Winston (2004), Sec. 9.2.

In a similar way, equation (9) and equations (10) determine, respectively, the benefit of increased fish abundance B^{abund} and mean density D_j in each subnetwork j. Note that the additional multiplier 120 in equation (9) is included to determine the total abundance of fish within a 120m stretch of river to match the assumption of the DCE. For equation (10), fish density is assumed to be some function $g(\cdot)$ of connectivity status $C_j^{d_2}$ evaluated at a dispersal distance d_2 and a vector of additional environmental covariates $\boldsymbol{\mu}_j$ influencing fish density. For our purposes, we used equation (5) to derive the following:

$$D_j = \gamma_0 + \gamma_1' C_j^{\text{0.1km}} \qquad \forall j \in J \tag{16}$$

with $\gamma_1' = \gamma_1[\overline{RUNS \times WIDTH}]^2$ and $\overline{RUNS \times WIDTH}$ being the mean of survey runs times survey width across all survey sites in our data set.

To continue, equations (11) determine the C metric connectivity of each subnetwork j at dispersal distances d_1 and d_2. Full details for working out the C metric are presented in Diebel et al. (2015). It can be shown that the C metric takes the following general form:

$$C_j^d = \sum_{k\in J|k<j} w_{jk} z_{kj} + \sum_{k\in J|k\geq j} w_{jk} z_{jk} \qquad \forall j \in J, d \in \{d_1, d_2\} \tag{17}$$

$$z_{jk} = \prod_{\ell\in B_{jk}} \left(p_\ell^0 + \sum_{i\in S_\ell} p_{\ell i}' x_{\ell i} \right) \qquad \forall j, k \in J \mid k \geq j \tag{18}$$

In equations (17)-(18), parameter w_{jk} represents the amount of habitat contributed by subnetwork k to subnetwork j, B_{jk} is the set of barriers along the path from origin subnetwork j to destination subnetwork

[7] For example, the UK government advocates a social time preference rate of 3.5% real for environmental projects (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/685903/The_Green_Book.pdf).

k, and variable z_{jk} specifies the cumulative passability between subnetworks j and k (i.e., the product of barrier passabilities in set B_{jk}). To linearize (18), one can use the probability chain method of O'Hanley et al. (2013) as demonstrated in King et al. (2017).

Finally, inequality (12) is a budget constraint on the total cost of barrier mitigation, inequalities (13) specify that at most one mitigation project can be carried out at each barrier j, and constraints (14) place binary restrictions on the barrier mitigation decision variables.

2.5 River Barrier Data

Spatial locations of 805 artificial in-stream structures in the River Wey catchment were derived by merging several existing UK barrier data sets. In addition, we also employed river cross-section and longitudinal profiles of the River Wey obtained from the EA. Barriers were snapped (50m snapping distance) onto an edited version of the EA's detailed river network (DRN) containing 1160km of waterway. In all, 669 structures where successfully snapped to the DRN.

To assess the passability of structures, a coarse resolution, rapid barrier assessment protocol (SNIFFER, 2010) was carried out at a sample (n = 63) of structures using on a combination of in-field measurements and photographic analysis. Criteria for assigning upstream and downstream passabilities in the continuous range 0 (impassable) to 1 (fully passable) to different structural types are described in King et al. (2017). For structures not directly assessed (n = 606), upstream/downstream passabilities were set to the median values for each structure type.

A single mitigation project was considered for each potential barrier with current upstream/downstream passability less than 1 (n = 650). Barriers located in the upper reaches of the catchment were considered suitable for complete removal or, in the case of culverts, replacement, which was assumed to restore full passability in the upstream and downstream directions. For barriers located in the middle and lower portions of the river network, removal was not considered feasible due to the need to maintain water levels for navigation. Instead, these barriers were considered candidates for the provision a fish pass that (optimistically) provided full downstream passability and 0.75 upstream passability. For locks, combined upstream/downstream was assumed to increase to 0.65 from improved and more regular operation.

The costs of barrier mitigation were estimated on the basis of information provided by the River Restoration Council for works at similar structures and information published by the EA (EA, 2010). The cost of mitigating all 650 candidate barriers is estimated at £53.3 million. The large magnitude of river barrier mitigation costs has previously been noted in the literature by Logar et al. (2019). However, while the cost of mitigating all barriers in the Wey may appear high, Grabowski et al. (2018) report that this is significantly less than projected costs to rehabilitate dams in the US.

3 Results

3.1 Willingness-to-Pay Results

We begin by reporting the WTP results for the River Wey which are summarized in Table 1. WTP estimates for both fish species richness and fish abundance are positive and statistically significant at the 0.01 level,

Table 1: Willingness-to-pay for River Wey ecosystem attributes.

	Fish Species Richness		Fish Abundance	
	WTP (£)	95% CI	WTP (£)	95% CI
River Wey	2.882***	2.174 – 3.589	0.099***	0.053 – 0.145

* p≤0.1, ** p≤0.05, *** p≤0.01

indicating residents of the Wey derive measurable benefit from these ecosystem attributes. Although not reported here, we note that a significant negative ASC was also found, implying that respondents may be willing to pay for river barrier mitigation for reasons other than improving the health of the local fish community. As such, the WTP reported reflect conservative estimates of respondents overall stated WTP for a program of barrier mitigation actions in the Wey.

3.2 Prediction of Fish Species Richness and Abundance Results

Fish species richness regression model results are summarized in Table 2. Parameter estimates for the intercept, $C^{10\text{km}}$, and $\sqrt{USL}$ are all statistically significant at the 0.01 level for the generalized Poisson model. The estimate for $RUNS \times AREA$ is statistically significant at the 0.1 level. A likelihood ratio test for the generalized Poisson regression confirmed that the explanatory variables are jointly significant at the 0.01 level. Note that for a Poisson model, the regression model parameters indicate the effect of a one unit increase for each explanatory variable on the logarithm of the dependent variable (i.e., the expected number of absent species $\ln(\bar{R})$).

To gain a better intuition of how the explanatory variables directly influence the expected number of absent species $\bar{R}$, we also report the marginal effects for the explanatory variables evaluated at the mean of the sample data. The marginal effect for $C^{10\text{km}}$ is also statistically significant at the 0.01 level and large relative to the observed mean number of absent species at survey sites (12.4 species), indicating that potentially large reductions (gains) in species absence (richness) can be achieved with increased network-scale connectivity. The scale parameter θ is negative and statistically significant at the 0.01 level, confirming underdispersion of the data.

With a pseudo R^2 of 0.472, measured as the square of the correlation between observed and predicted number of absent species (Eisenhauer, 2003), the Poisson model accounts for roughly half the variation of the dependent variable, slightly better than the OLS model with a pseudo R^2 of 0.468. The better fit of the Poisson model is also confirmed by the lower value for the Akaike information criterion (AIC) – 496 for the Poisson model compared to 514 for the OLS model.

The results of the fish abundance regression model are reported in Table3. There is a strong positive relationship between connectivity and fish density, with parameter estimates all significant at the 0.1% level. In addition, the marginal effect for $C^{0.1\text{km}}$ is both significant at the 0.1% level and higher than observed mean density at survey sites (1.45 fish/m), indicating that increased local connectivity could lead to a sizable increase in fish abundance. The R^2 of the model is 0.643, further indicating that the model has generally good predictive ability accounting for over 60% of explained variance.

Table 2: Regression model results for predicting fish species absence in the River Wey.

Parameter	OLS		Generalized Poisson			
	Est.	SE	Est.	SE	Marginal Effect	SE
α_0	2.864***	4.594×10^{-2}	2.882***	4.097×10^{-2}	–	–
α_1	-1.800***	0.550	-1.752***	0.526	-21.507***	6.472
α_2	-3.029×10^{-4}***	5.833×10^{-5}	-3.9×10^{-4}***	6.176×10^{-5}	-4.82×10^{-3}***	7.3×10^{-4}
α_3	-4.611×10^{-5}***	1.434×10^{-5}	-2.953×10^{-5}*	1.626×10^{-5}	-3.6×10^{-4}*	2.0×10^{-4}
θ	–	–	-3.775×10^{-2}***	3.44×10^{-3}	–	–
R^2	0.444		–			
pseudo R^2	0.468		0.472			
AIC	514		496			

* p≤0.1, ** p≤0.05, *** p≤0.01

Table 3: Regression model results for predicting fish abundance in the River Wey.

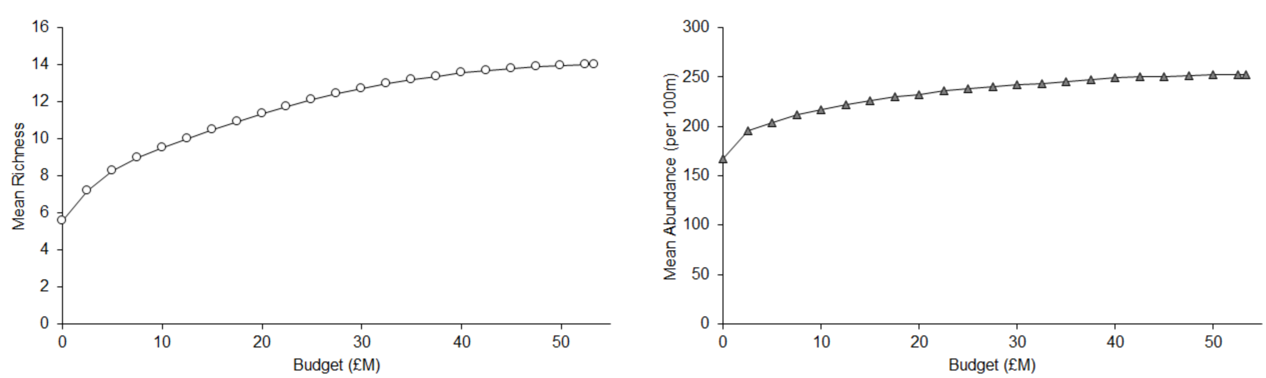
Parameter	OLS			
	Est.	SE	Marginal Effect	SE
γ_0	0.506***	0.130	–	–
γ_1†	4.324×10^{-3}***	5.692×10^{-4}	1.649***	0.217
R^2	0.643			
AIC	64			

*** p ≤0.001

†Independent variable $C^{0.1\text{km}}$ only

3.3 Bioeconomic Model Results

The results for our bioeconomic model are presented in Figures 3 and 4. To generate these results, we had to make an assumption regarding the size of the population who will benefit from the mitigation activities. Specifically, we assumed that the benefiting population was drawn from postcodes local to the Wey. Based on the number of 'all usual resident' counts for each postcode as recorded in the 2011 UK national census and accessed via the Office for National Statistics NOMIS website (ONS, 2013), we estimated the number of individuals at 881,033. This was then converted to 367,097 households using the national average of 2.4 persons per household (ONS, 2012).



(a) (b)

Figure 3: Fish richness (a) and abundance (b) as functions of mitigation budget.

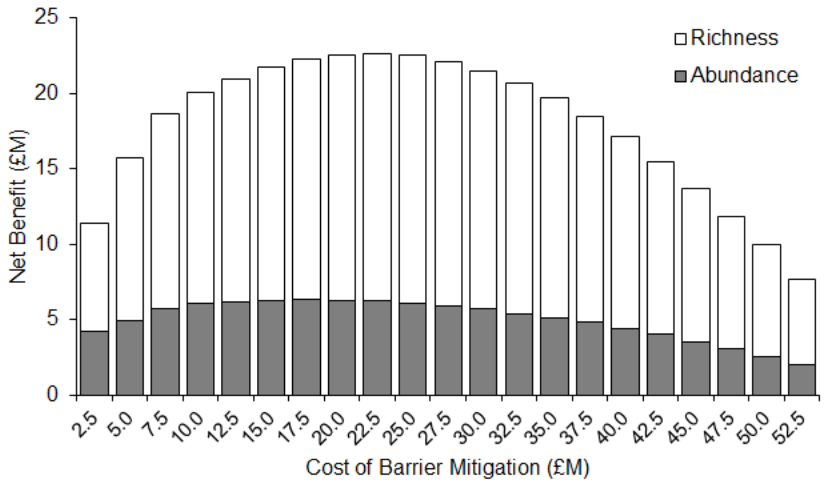


Figure 4: Fish richness (a) and abundance (b) as functions of total mitigation cost.

The results show clear decreasing marginal returns for both increased fish richness and abundance with increases in the barrier mitigation budget (see Figure 3). For example, mean richness increases by 71% from approximately 5.6 species to 9.5 species (net gain 4.0 species) given a mitigation budget of £10M. Increasing the budget from £10M to £20M, however, only produces a net gain of 1.8 species or an additional 33% increase in richness. Similarly, mean abundance increases by 30% (from 167.4 to 217.3 fish/m) for the first £10M allocated to barrier mitigation, but only by an additional 9% (from 217.3 to 232.5 fish/m) given an additional £10M. The overall pattern of a concave shaped response curve for restoration gain (typically measured as accessible habitat) versus budget has been observed in various other studies on barrier mitigation planning (Kuby et al., 2005; O'Hanley and Tomberlin, 2005; Zheng et al., 2009). These findings reveal that substantial ecological gains can be achieved with even modest investment in river barrier mitigation.

It is worth noting here that the fish species richness and abundance curves plateau at different budget thresholds. Increases in richness quickly level off after the budget reaches around £30 to 35M. Abundance, meanwhile, levels off at a much lower budget, roughly £20 to 25M. This makes intuitive sense, since the C metric will tend to approach its maximum more rapidly as river subnetworks are reconnected when the dispersal distance is small (i.e., for abundance) compared to when it is large (i.e., for richness). The shape of the richness and abundance curves has implications for understanding the economic value of barrier mitigation, as the benefits derived from increased richness and abundance as a function of budget will perfectly mirror the richness and abundance versus budget curves (i.e., they are simply rescaled by their respective WTP estimates).

Analysis of the economic benefits of barrier mitigation (see Figure 4) reveals that net benefit quickly rises with increased budget, reaching a peak of £22.6M at a budget of £22.5M, and steadily decline thereafter. As the net benefit curve is quite flat for budgets between approximately £17.5M and £27.5M, one could justifiably argue that investment in barrier mitigation anywhere within this range is economically efficient. However, from a purely cost-benefit perspective, it could also be argued that only a few or even all barriers in the Wey should be mitigated, as net benefits are positive ($\geq$ £6.8M) for all nontrivial budgets considered (£2.5 to 53.4M).

Looking more closely at the breakdown of net benefits, species richness accounts for by far the largest contribution. On average, net benefits derived from increased species richness range from 1.7 to 2.9 times higher than net benefits derived from increased abundance. At the optimal barrier mitigation budget of £22.5M, mean abundance increases by 41% (+69.2 fish/m) and mean richness by 111% (+6.1 species) compared to the current baseline, resulting in net benefits of £6.3M for abundance and £16.3M for richness (approximately 2.6 times higher). The fact that net benefits derived from increased richness are higher than net benefits derived from increased abundance is in part down to the much larger WTP for richness relative to the WTP for abundance (see Table 1) and part due to the shape of the richness and abundance versus budget curves (see Figure 3).

4 Discussion and Conclusions

There continues to be increasing interest among river managers and policy makers to remove or mitigate artificial barriers in order to reduce river fragmentation and enhance the ecological integrity of fluvial ecosystems. Besides aligning with goals to protect freshwater biodiversity,[8]this interest also stems from a desire to improve river ecosystem function and the supply of ecosystem services they provide (NEA, 2011; Rounsevell et al., 2018). In this paper, we present an approach to cost-effectively prioritize barrier mitigation actions to maximize restoration gains and estimate the economic benefits of barrier mitigation for the purposes of undertaking a CBA. We achieve this by combining a DCE to estimate WTP for increased fish species richness and abundance with the specification and parameterization of empirical models of fish population responses to barrier mitigation into an MILP bioeconomic model. The bioeconomic model produces an optimized portfolio of barrier mitigation decisions that maximize fish species richness and abundance gains given a limited budget. Integrating WTP estimates derived from a DCE subsequently allows us to examine the net benefits of barrier mitigation based on the increases in river ecosystem services derived from these biophysical attributes. We demonstrate our integrated modeling approach using data from the River Wey catchment in southeast England. Our results indicate that implementation of a barrier mitigation program in the River Wey would be beneficial for any level of investment and economically efficient given an expenditure of £22.5 million, the socially optimal level of investment in barrier mitigation activity (i.e., where marginal costs equal marginal benefits).

The relevance of our methodology is that CBA of environmental policy is now assumed to be carried out by matter of course by environmental agencies, for example under government rule making in the US and the WFD in the EU (Johnston and Rosenberger, 2010). However, as noted previously, the actual extent and frequency of CBA is limited in practice (Berbel and Expósito, 2018; Logar et al., 2019), which in part can be traced to a lack of appropriately developed analytical frameworks that are suited to interdisciplinary analysis. Consequently, it is anticipated our integrated modeling approach will be of direct benefit to both policy makers and practitioners involved in river ecosystem management and barrier mitigation. Our methodological approach, as illustrated by our case study, allows for the identification of levels of investment that deliver high social benefits at costs that can be justified in the policy context. This is likely to be particularly relevant to EU member states developing river basin management plans (RBMPs) to be adopted in 2021, both from the perspective of delivering obligations under the WFD, as well as the EU Biodiversity Strategy commitment to restore 25,000km rivers to be free-flowing. Despite Brexit, our framework is also highly relevant to the

[8]Changes in freshwater ecosystem connectivity is likely to be a suggested monitoring element under Target 1 of the UN Convention on Biological Diversity Post-2020 Global Biodiversity Monitoring Framework (https://www.cbd.int/doc/c/2c69/df5a/01ee87752c3612d3ba7ec341/wg2020-02-03-add1-en.pdf)

UK given the government's commitments under the 25 Year Environment Plan to restore waters to be close to their natural state and to exceed the objectives of RBMPs developed under the WFD (Defra, 2020).

Another feature of our analysis is that it reveals when costs of river restoration are excessive compared to benefits. In cases such, analysis can be used to examine applications for derogations (i.e., exemptions) from achieving ecological targets within the timescales set out in water policies such as the WFD. Likewise, it is also the case that by examining river restoration projects using a robust methodological framework, some of the concerns regarding the excessive use of derogations could be avoid. As noted by Boeuf et al. (2018) with regard to the WFD, there has been a great deal of ambiguity in terms of how derogations have been implemented, which can in part can be traced to how different countries select and implement analytical methods used to evaluate exemptions. As our framework is transparent and flexible, it allows tailor made and Pareto efficient restoration plans to developed on a watershed by watershed basis, thereby avoiding situations where derogation is supported based on analysis of the economic efficiency of one-size-fits-all restoration policies developed *a priori*.

Looking to the future, the need to develop sound integrated modeling frameworks will increase as potentially new policy trade-offs emerge in relation to rivers. Specifically, there is already growing pressure within the UK and EU to explore and increase the generation of green electricity via hydropower schemes. Within the EU, there are some 25,000 existing hydropower plants (EEA, 2015). However, it is far from clear if reliance on hydropower plants to achieve EU renewable energy targets is compatible with the WFD or Biodiversity Strategy. It is issues such as this that the methodology presented in this paper can help address by examining and identifying key trade-offs that are going to need to be made.

Finally, whilst we believe our case study provides significant insight into river barrier mitigation issues, we stress that the analysis presented here is meant for illustrative purposes only. Our economic analysis is considered reasonably robust with respect to potential variations in WTP and the benefiting population selected. However, considerable uncertainty exists with respect to estimation of the fish species richness-connectivity response parameter (α_1 in equation (4)) and to a lesser extent the fish abundance-connectivity parameter (γ_1 in equation (5)). There are a variety of reasons to expect uncertainty with respect to these parameter. For instance, while animal population sampling data is often characterized by significant variability due the dynamic behaviors of animal populations in both space and time (Link et al., 1994), the amount of useable fish survey data is somewhat limited and it is unclear if survey locations and dates were systematically determined by the EA or simply convenience surveys. More importantly, barrier inventorying that was undertaken was not supported by a full-scale survey of the River Wey catchment. Consequently, the barrier dataset may be incomplete. Furthermore, only about 20% of barriers identified underwent in-field assessment, with passabilities for the remainder of barriers inferred from those of the same structural type. In any real-world application, the quality of solutions to the bioeconomic model would be much improved with provision of a more comprehensive barrier inventory and assessment of barrier passability and a fish population survey specifically designed to inform the analysis.

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