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A Unified Model for Optimizing Riverscape Conservation

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Abstract

1. Spatial prioritization tools provide a means of finding efficient trade-offs between biodiversity protection and the delivery of ecosystem services. Although a large number of prioritization approaches have been proposed, most are specifically designed for terrestrial systems. When applied to river ecosystems, they often fail to adequately account for the essential role that landscape connectivity plays in maintaining both biodiversity and ecosystem services. This is particularly true of longitudinal connectivity, which in many river catchments is highly altered by the presence of dams, stream-road crossings, and other artificial structures.
2. We propose a novel framework for coordinating river conservation and connectivity restoration. We formulate an optimization model for deciding which subcatchments to designate for ecosystem services and which to include in a river protected area (RPA) network, while also deciding which existing river barriers to remove in order to maximize longitudinal connectivity within the RPA network. In addition to constraints on the size and makeup of the RPA network, the model also considers the suitability of sites for conservation, based on a biological integrity index, and connectivity to multiple habitat types. We demonstrate the usefulness of our approach using a case study involving four managed river catchments located in Hungary.
3. Results show that large increases in connectivity-weighted habitat can be achieved through targeted selection of barrier removals and that the benefits of barrier removal are strongly depend on RPA network size. We find that (i) highly suboptimal solutions are produced if habitat conservation planning and connectivity restoration are done separately and (ii) RPA acquisition provides substantially greater marginal benefits than barrier removal given limited resources.

4. Synthesis and applications. Finding a balance between conservation and ecosystem services provision should give more consideration to connectivity restoration planning, especially in multi-use riverscapes. We present the first modelling framework to directly integrate and optimize river conservation and connectivity restoration planning. This framework can help conservation managers to better account for connectivity, resulting in more effective catchment scale maintenance of biological integrity and ecosystem services delivery.

Introduction

One of the greatest challenges facing society today is the urgent need to halt the global decline of biodiversity, while maintaining the capacity of ecosystem services for human well-being (Bennett et al., 2015). Various studies have investigated the complex relationship between biodiversity and ecosystem services (Reyers et al., 2012; Howe et al., 2014). Ideally, management actions should be designed to provide a wide range of benefits, both in terms of conservation and ecosystem services (a win-win situation). Often, increased biodiversity conservation can only be achieved at the loss of certain ecosystem services and vice versa (a win-lose situation). This is frequently the case in heavily used, human dominated landscapes, where environmental managers must make difficult choices between biodiversity and ecosystem service provision (Palomo et al., 2014).

A potential solution to this dilemma is to try to maximize the number of win-win and decrease the number of win-lose situations by using spatial prioritization to find the best trade-off between biodiversity protection and the delivery of ecosystem services (Cordingley et al., 2016; Doody et al., 2016). Such approaches, however, are still uncommon in practice. Most spatial prioritization methods focus on the delineation of ecosystem service hotspots (i.e., by selecting areas that are high in value for one or sometimes multiple services), rather than explore potential conflicts and synergies between biodiversity and ecosystem services (Cimon-Morin et al., 2013; Schröter & Remme, 2016).

Looking specifically at prioritization in riverine ecosystems, a frequently neglected consideration is the critical role that landscape connectivity plays in the maintenance of both biodiversity and ecosystem services (Taylor et al., 1993; Mitchell et al., 2013). Rivers provide a multitude of vital ecosystem services, such as water supply, navigation, hydropower, fishing, and recreational opportunities (Vörösmarty et al., 2010). Many of these services are dependent on basic ecosystem processes, including species movements, genetic exchange, and material and energy flows, which are all strongly regulated by longitudinal connectivity. At the same time, the dendritic structure of rivers makes them particularly susceptible to connectivity disruption (Grant et al., 2007; Hermoso et al., 2011), which, in turn, can adversely impact ecosystem integrity. Indeed, river ecosystems are among the most threatened worldwide, in large part because of the presence of large numbers of dams, stream-road crossings, and other hydromodifications (Dynesius & Nilsson, 1994; Januchowski-Hartley et al., 2013).

To date, research on prioritizing river habitat protection and connectivity restoration actions has progressed mostly along two separate paths. One line of enquiry concerns the development of planning tools for prioritizing the repair/replacement/removal (i.e., mitigation) of artificial river barriers that impede aquatic organism passage, mainly fish, using graph theory and optimization techniques (Erős et al., 2011; Neeson et al., 2015; King et al., 2017). A separate strand of research has focused on applying reserve selection methods (Moilanen et al., 2008; Newbold & Siikamäki, 2009; Linke et al., 2012, Hermoso et al., 2017) to the design of freshwater conservation networks. Within this latter group, connectivity, when it has been considered, is incorporated in a fairly simplistic manner by trying to ensure that selected areas (usually subcatchments) are spatially adjacent. In neither of these two research themes has the potential presence of instream barriers and their associated impacts on longitudinal connectivity been addressed together with conservation planning.

In this study, we address this shortcoming by proposing a novel approach to systematic river conservation and connectivity restoration planning. More specifically, we formulate a model for jointly optimizing the selection of river protected areas and barrier removals. Given a set of biodiversity elements (i.e., habitat classes) in need of conservation, the aim of the model is to maximize longitudinal connectivity between selected areas through targeted barrier removals, subject to lower/upper limits on the amounts of protected habitat and a cap on the number of barrier removals. The model adopts a limiting factors approach, in which connectivity of any given river protected area is based on the minimum level of connectivity to any other habitat class. We subsequently demonstrate the usefulness of our model using a case study involving four river catchments located in Hungary.

Underpinning our optimization model is a conceptual model (Fig. 1) that provides general guidelines on how to systematically plan out management actions in the context of biodiversity protection and ecosystem services delivery. The conceptual model combines three main steps: 1) establishment of biodiversity and ecosystem service indicators; 2) definition of a suitable connectivity metric; and 3) application of a spatially explicit prioritization approach to efficiently allocate land use and connectivity restoration management actions.

The first step is to develop a set of “indicators” of biodiversity and ecosystem services, namely the key biological/physical elements of a system that help to maintain biodiversity and ecosystem services and the various pressures that degrade ecosystem structure and function (Grizetti et al., 2016; Maes et al., 2016). For example, physical and chemical water quality, land use type, invasive species threats, and the presence of in-stream barriers can provide useful indicators of overall ecosystem health in freshwaters (Nelson et al., 2009, Terrado et al., 2016; Vital-Abarca et al., 2016).

The next step is to assess the role of connectivity in relation to biodiversity and ecosystem services regulation in a particular system and to propose a metric that adequately describes connectivity. An important consideration is the role of connectivity in producing trade-offs between biodiversity and various ecosystem services. Although connectivity is critical for the structuring and functioning of natural ecosystems, its importance to the delivery of ecosystem services varies greatly. In stream ecosystems, for example, connectivity is critically important for the dispersal of fish species, which are key components of ecosystem function and provide various ecosystem services (e.g., recreational and commercial fishing, aesthetic value, see Holmlund & Hammer, 1999). On the contrary, connectivity may be less important for the provision of urban/agricultural water supply or for electricity, where, in fact, the damming of rivers is the main way these are supplied (Auerbach et al., 2014; Grizetti et al., 2016).

With regard to the choice of a suitable connectivity metric, this depends on basic characteristics of the system. In terrestrial applications, the adjacency/compactness of spatial units makes intuitive sense (McDonnell et al., 2002; Nalle et al., 2002). In riverine systems, however, connectivity between two different points in a river is dictated by the river's flow paths, making indices like the Dendritic Connectivity Index (Cote et al., 2009), which take into account the passability of in-stream barriers, much more suitable (Erős et al., 2012).

Lastly, because resources for conservation and connectivity restoration are limited, it is essential for landscape management to allocate resources in the most efficient way possible. The recommendation to use a spatially explicit prioritization approach leaves two reasonable alternatives: graph theory models (Erős et al., 2011) and optimization models (King et al., 2017). Optimization has the distinct advantage over graph theory in being prescriptive rather than descriptive (King & O'Hanley, 2016), meaning that is produces a recommended course of action that aims for the best allocation of limited resources to maximize benefits (i.e., biggest bang for the buck). Moreover, optimization models are perfectly suited to balancing multiple, potentially competing goals, thus making them ideal for driving negotiation among decision makers and delivering more win-win scenarios that promote biodiversity protection and ecosystem services provision.

Materials and Methods

Study Area

We selected four river catchments located in Hungary for our study (Fig. 2). These include Lake Balaton (5775 km^2), the Marcal River (3084 km^2), the Sajó River (5545 km^2), and the Zagyva River (5677 km^2). Catchments differ considerably in terms of the mix of land uses, stream habitat type, and number of artificial barriers present (Tab. 1). The dominant land cover type is agricultural (mainly arable land, vineyards to a smaller extent), but deciduous forests, pastures, grasslands, and wetlands are also present. Urbanization is primarily confined to small cities and villages. River habitat can be categorized into five broad types: lowland river, lowland stream, highland river, highland stream, and submontane stream (Erős, 2007).

Biodiversity and Ecosystem Services Indicators

Conservation area selection methods often use simple biological diversity indicators as proxies of conservation value (e.g., richness, species occurrences, endemism, species composition). Rarely is attention given to the biological integrity of the ecosystem, even though this may be a better indicator of a particular location’s value for conservation purposes (Angermeier & Karr, 1994; Karr, 1999; Peipoch et al., 2015). According to Angermeier and Karr (1994), “diversity is a collective property of system elements, integrity is a synthetic property of the system.” Diversity quantifies the variety of items in the system (e.g., species richness, number of functional forms), whereas integrity refers to the number of components (diversity) and the processes that contribute to the continued functioning of the system in a natural state. In this sense, integrity emphasizes the degree to which a system has been altered from its natural (i.e., undisturbed) state (Hawkins et al., 2000; Pont et al., 2006). An ecosystem with high integrity indicates that natural ecological, evolutionary, and biogeographic processes are intact (Angermeier & Karr 1994; Angermeier 2000; Beechie et al., 2010). Although biodiversity and biological integrity are often confused, it is important to distinguish between the two, especially in the context of examining biodiversity/integrity and ecosystem service relationships. For example, a reservoir created by the presence of a dam may have higher biodiversity than a free-flowing stretch of river because of the occurrence of both lotic and lentic species (especially waterbirds and macrophytes, which are normally less abundant in undisturbed lotic areas). Stream segments impounded by a reservoir can also be valuable for the provision of ecosystem services (e.g., water storage/withdrawal and recreational fishing), but clearly have lower biological integrity compared to natural stream segments (Beechie et al., 2010; Thorp et al., 2010; Auerbach et al., 2014).

We quantifid the biological integrity of stream segments and their associated subcatchments using five indicators of conservation quality and naturalness. These include: 1) land use intensity; 2) absolute conservation value for fish fauna; 3) relative conservation value for fish fauna; 4) biological integrity of fish fauna; and 5) biological water quality. Land cover categories are important indicators of ecosystem services (Grizetti et al., 2016; Maes et al., 2016). In this study, we used the land use index (LUI) of Böhmer et al. (2004), which describes land use intensity and impact within a catchment along a gradient from natural forest cover to agricultural and urban use. The index, which has been used in other studies (e.g., Ligeiro et al., 2013), is calculated as follows:

$$\text{LUI} = \%\ \text{pasture} + 2 \times \%\ \text{arable land} + 4 \times \%\ \text{urban area}$$

Fish assemblages are frequently used for selecting conservation areas in riverine ecosystems (Filipe et al., 2004; Sowa et al., 2007). Fish are also an important focus for river connectivity restoration. The absolute (ACV) and relative (RCV) conservational value of fish fauna in each stream segment was determined using the index of Antal et al. (2015). To calculate ACV, increasing weights were assigned to fish taxa according to their extinction risk as follows:

$$\text{ACV} = 6n_{\text{EW}} + 5n_{\text{CR}} + 4n_{\text{EN}} + 3n_{\text{VU}} + 2n_{\text{NT}} + n_{\text{LC}}$$

Here, n_{EW} is the number of extinct species in the wild, n_{CR} is the number of critically endangered species, n_{EN} is the number of endangered species, n_{VU} is the number of vulnerable species, n_{NT} is the number of near threatened species, and n_{LC} is the number of least concern species (see Erős et al., 2011, Antal et al., 2015). To calculate RCV, the absolute value was divided by the total number of species. Similar approaches for other taxonomic groups can be found in the literature (Fattorini, 2006).

Biological integrity of fish assemblages (BIF) was determined using the method of Sály and Erős (2016). BIF quantifies the degree of alteration of fish assemblages compared to near-natural (reference) fish assemblages based on the structural and functional properties of the fish fauna and their responses to different stressors (i.e., land use, water quality, and hydromorphological alteration). Conceptually, BIF is similar to many other fish based biotic indices (Roset et al., 2007). Additional information about how BIF was determined are provided in an online appendix (see Appendix S1, Supporting Information).

Biological water quality (BWQ) is an integrative measure of the overall quality of the water for biota. Following procedures established by the EU Water Framework Directive, biological water quality was determined using the worst quality class value of five biological quality indices, which measure biological water quality based on the taxonomic and functional structure of benthic and water column algae, macrophytes, macroinvertebrates, and fish (Birk et al., 2012). Further details about BWQ are discussed in an online appendix (see Appendix S1, Supporting Information).

All five indices (LUI, ACV, RCV, BIF, and BWQ) were measured on a 5-point scale. An aggregate biological integrity index (BII) was then determined for each stream segment by taking the median of the five indices. Stream segments with high biological integrity scores represent locations with higher biodiversity conservation value. They are also essential for various regulatory (e.g., natural nursery areas) and cultural (e.g., recreational hiking) ecosystem services (Grizetti et al., 2016; Vital-Abarca et al., 2016).

Besides the quantification of biological integrity, we also used several pressure indices to identify areas within the river networks that may be better suited for alternative uses other than conservation and connectivity restoration. This includes subcatchments with a high urban/agricultural land use index and those where fish ponds, reservoirs, and waste water treatment plants are present. Such areas are often primarily devoted to agriculture/aquaculture, recreational fishing, flood control, or other ecosystem service uses and usually have low biological integrity anyway (a clear win-lose situation). Based on this initial screening process, all subcatchments deemed unsuitable for conservation/connectivity restoration a priori were assigned a BII value of zero (Fig. 2).

Barrier Survey Data

Barrier locations were extracted from a geo-database developed by the National Water Authority of Hungary. The database includes GPS referenced location information, structure type (e.g., dam, road crossing, sluice), and binary passability values of potential artificial barriers to fish movements. During field surveys, we further refined and updated this database for the four catchments in our case study during the summer and autumn of 2016 (July to November). We verified the exact location of barriers (Fig. 2), measured basic structural data, and estimated upstream-downstream passability. A road network map was also used to identify the location of bridges and estimate passability values for this type of barrier. In the field, we determined for each barrier its height, length, and slope, type (e.g., sluice, weir, dam, culvert, bridge), primary construction material (e.g., concrete, rock with concrete), internal/overflow water velocity, and substrate percentages (rock, stone, gravel, sand, silt, and concrete) both downstream and upstream of the barrier "wall."

To estimate upstream barrier passabilities for adult cyprinids (the dominant fish species in our study area), we used the rapid barrier assessment methodology described in King et al. (2017). Passability represents the fraction of fish (in the range 0-1) that are able to successfully negotiate a barrier in a particular direction. Each barrier assessed in the field (n = 703) was assigned one of four passability levels: 0 if a complete barrier to movement; 0.3 if a high-impact partial barrier, passable to a small portion of fish or only for short periods of time; 0.6 if a low-impact partial barrier, passable to a high portion of fish or for long periods of time; and 1 if a fully passable structure (these latter structures were subsequently excluded from analysis). We estimated adult cyprinid passability under both normal flow conditions and bankfull width conditions. Bankfull width levels were clearly visible from the shape of the channel and the location of riparian vegetation (Gordon et al., 1992). For barriers that could not be surveyed because of logistical difficulties (n = 101), we assigned the median passability values for a given barrier type.

Our surveys revealed the dominant types of barriers were stepped weirs, notched weirs (for flow measurement), small fishpond dams, large reservoir dams (for irrigation and water supply), and sluices. Contrary to many other countries (e.g., the US) where road culverts represent the main barrier type (Januchowski-Hartley et al., 2013), such barriers are relatively rare across Hungary (<1% of barriers surveyed). We also found that passability estimates were very similar regardless of normal versus bankfull width flow conditions. Consequently, we used passabilities under normal flow conditions for assessing river connectivity. Further, given that 95% of surveyed bridges were fully passable, we excluded this type of barrier in our analysis.

River Protection and Connectivity Optimization Model

To design efficiently a river protected area (RPA) network, we developed a spatial optimization model to decide: 1) which subcatchments to include within the RPA network and

2) which barriers to mitigate (i.e., remove, repair, install with a fish pass, etc.) to maximize longitudinal connectivity of the RPA network. Unlike existing optimization based methods for designing RPA networks, conservation planning and connectivity restoration are made simultaneously and their interactive effects were accounted for within our model. Full mathematical details of the model are provided in an online appendix (see Appendix S2, Supporting Information).

In brief, we assume that a study area is composed of one or more large, self-contained catchments, with each catchment made up of potentially multiple subcatchments. Any spatial resolution can be considered, from a few large subcatchments down to many small subcatchments. Although a subcatchment is the main selection unit, we do not necessarily assume that an entire subcatchment must be fully protected, just the river segments within a selected subcatchment. The conservation value of river segments is based on a weighted combination of the amount of habitat (i.e., length) and biological integrity (i.e., BII).

Longitudinal connectivity is quantified using a novel extension of the dendritic connectivity index (DCI) proposed by Cote et al. (2009). More specifically, we evaluate DCI at the local, segment-level scale (Mahlum et al. 2014) separately for each habitat type (lowland river, lowland stream, highland river, highland stream, and submontane stream) and then take the minimum value as an overall measure of segment connectivity. In this way, our model adopts a "limiting factors" approach by focusing on the habitat type in shortest supply.

There are a number of constraints considered within the model for modifying the size and makeup of the RPA network. These include:

(i) An upper limit on the size of the RPA network (i.e., the RPA network must be less than or equal to some fraction of available river habitat).

(ii) There must be a certain mix of habitat types within the RPA network (i.e., the fraction of each river habitat type must be greater than or equal to a specified threshold).

(iii) A constraint on the number of barrier removals.

For our case study, we considered two barrier mitigation options: 1) full barrier removal, with passability restored to 1 and 2) partial barrier removal, with passability restored to 0.5 if passability currently less (Noonan et al., 2012). We assumed full removal was possible only if a barrier was located in the RPA network. For a barrier outside the RPA network, only partial removal was available under the presumption that the barrier was essential in providing other ecosystem services (e.g., irrigation and water supply).

Our basic model includes separate constraints for RPA size and number of barrier removals (constraints (i) and (iii) above). Given cost estimates for barrier removal and RPA land acquisition, these can be easily replaced by a single budget constraint on overall cost. To explore this option, a figure of €5000 per ha was used for RPA purchase (based on the cost of prime agriculture land), €400k for full barrier removal, and €200k for partial barrier removal. As the cost of acquiring an entire subcatchment is prohibitively expensive, we assumed that only riparian areas within a 30 m distance of selected river segments had to be purchased. Studies have indicated that ≥30 m buffer strips are generally sufficient to protect most aquatic species (Lee et al., 2004).

Results

BII values varied widely both within and among the catchments (Fig. 2). In general, the Balaton Catchment contained a high number of subcatchments with low or zero BII values, indicating that a large part of this catchment is not ideally suited for conservation but other land use functions instead. The Sajó Catchment, on the other hand, contained the highest number of subcatchments with high BII values.

Maximum connectivity-weighted habitat for different sized RPA networks varied as a function of the number of full/partial barrier removals (Fig. 3). Even with a small number of barrier removals, impressive gains in connectivity-weighted habitat could be achieved. For example, with a moderate sized RPA network comprising 40% of selectable river length ($\theta = 0.4$), connectivity-weighted habitat increased by more than 100% (from a baseline value of 1355.46 to 2813.28) when just 6 barriers were removed. In fact, strong diminishing returns were observed as the number of barrier removals increased, as indicated by the concaved shapes of the connectivity-weighted habitat versus barrier removal curves. Further, the benefits of barrier removal were proportional to the size of the RPA network. For example, for the smallest sized network encompassing 10% of selectable river length ($\theta = 0.1$), the removal of 4 barriers resulted in a 26% increase in connectivity-weighted habitat. In contrast, for a much larger sized network incorporating 60% of selectable river length ($\theta = 0.6$), the removal of 4 barriers resulted in a 132% increase in connectivity-weighted habitat.

To investigate how equitably protection resources are allocated among the different river catchments (Balaton, Marcal, Sajó, and Zagyva), we determined the fraction of the RPA network contained in each catchment for selected values of θ given no barrier removal versus an unrestricted number of barrier removals (Figs. 4 and 5). We found that both network size and barrier removals strongly influenced the spatial pattern of selected subcatchments. For the smallest sized reserve network ($\theta = 0.1$), protection resources are concentrated almost entirely in the Balaton (95%) regardless of whether barriers can be removed or not (Figs. 4a, 4b, and 5a). At the other extreme, the possibility of removing barriers also does not appear to dramatically alter the spatial distribution of the largest sized network ($\theta = 0.9$), with a much more even spread among catchments appearing with and without barrier removal. For the intermediated sized networks ($\theta = 0.3, 0.5, 0.7$), the pattern is more complex. Without barrier removals (Fig. 4a), the distribution of protected habitat among catchments becomes progressively more balanced with increasing RPA network size. With barrier removals (Fig. 4b), conservation resources are directed out of the Zagyva and Balaton and into the Marcal ($\theta = 0.3$) and then the Sajó ($\theta = 0.5, 0.7$; see also Fig. 5b).

The clear preference for concentrating conservation resources in the Balaton for the smallest sized RPA network is somewhat surprising given that it is one of the most well-developed areas in Hungary in terms of urbanization, aquaculture, and tourism and has a barrier density (number of barriers per length of river) more than double that of any other catchment (Tab. 1). Evidently, the Balaton is an ideal location for constructing an RPA network given limited conservation resources; it contains a significant proportion of three out of five habitats types (i.e., highland stream, lowland stream, and lowland river) and a particularly favorable arrangement of mostly well-connected river segments. The only way for the allocation of conservation resources to dramatically shift is by modifying the basic design of the RPA network (i.e., by adjusting the minimum percentage of each habitat type). Overall, the two least common habitats in the four catchments are submontane stream (5.6%) and lowland river (6.6%). Doubling the minimum fraction of these habitats from 80% to 160% (i.e., setting $\alpha = 1.6$ for these two habitat types and leaving the others at 0.8), the Balaton would account for a greatly reduced, albeit still high, share (59-64%) of the $\theta = 0.1$ sized RPA network (see Appendix S3, Supporting Information). Putting very high α weights on submontane streams and highland rivers, the two least common habitat types in the Balaton, would similarly reduce the amount of resources allocated to the Balaton (results not shown). These examples demonstrate the flexibility of the model with regard to finding alternative solutions that meet management needs. They also show that when optimizing limited conservation/restoration resources, rather counterintuitive results can sometimes be obtained. For example, each catchment contains roughly similar amounts of river length eligible for conservation (Tab. 1), with the Balaton, Marcal, Sajó, and Zagyva contributing 22%, 19%, 33%, and 26% of the total, respectively. Yet the fraction of river habitat conserved in each catchment can be very far from equal depending on the size of the RPA network and the barrier removal budget.

We also wanted to ascertain the importance of coordinating river protection and barrier removal decisions. There is considerable variability in relative connectivity-weighted habitat gain when river protection decisions are made first and barrier removal decisions second (Fig. 6). Note that solutions for $b = 0$ and $\theta = 1$ are not considered, as these will always be optimal using a two-stage approach. Results showed that river protection and restoration decisions are strongly interdependent (Fig. 6). By optimizing barrier removal decisions separately from river protection decisions, far less connectivity-weighted habitat is obtained, with the effect exacerbated as the size of the reserve network increases. For smaller sized networks ($0.1 \leq \theta \leq 0.3$), 68-91% of maximum connectivity-weighted habitat can be achieved (interquartile range) across all barrier removal scenarios. For moderate and large sized networks ($0.4 \leq \theta \leq 0.9$), however, the opportunity cost of sequential decision making are much higher, with only 57-76% of the maximum being achieved (interquartile range). In the worst case, just 52% of the maximum is achieved, demonstrating that highly suboptimal solutions may be obtained if river protection and connectivity restoration decisions are not properly coordinated.

Lastly, we wanted to examine the relative effectiveness of barrier mitigation against RPA land purchases. To do this, we modified our basic model by first including estimates for barrier removal and land purchase costs and then used a single budget for overall cost (in place of separate budgets for land acquisition and barrier removal). Connectivity-weighted habitat increased in a roughly linear fashion with budget (Fig. 7a). This differed from the strong diminishing returns observed for our basic model with fixed RPA size and an increasing number of barrier removals (Fig. 3). RPA land purchases made up the majority of total spend regardless of budget (Fig. 7b). At lower budgets (€5-30M), RPA land purchases accounted for up to 93% of total cost. As budget increased, this percentage decreased but never below 73% of total cost (at €100M). These results suggest that RPA acquisition provide substantially greater marginal benefits than barrier removal, especially if resources are limited.

Discussion

In this study, we demonstrate the benefits of combining river protection and connectivity restoration planning in multi-use riverscapes. As with other related work (Doody et al., 2016; Zheng et al., 2016), our framework recognizes the need for a spatially informed and strategic approach to the selection of different land uses for the catchment level delivery of biodiversity protection and ecosystem services. Our framework is noteworthy in being the first to directly incorporate connectivity restoration planning into the prioritization process using an optimization based approach. Our methodology attempts to unify systematic reserve selection planning with connectivity restoration planning, thus providing a powerful tool to help guide protection of river ecosystems. Optimization approaches, such as ours, are specifically designed to find the best allocation of limited resources to achieve one or more planning goals. They are also useful for generating Pareto optimal trade-off curves, which can reveal how conservation and other objectives vary with different levels of investment (Neeson et al., 2015).

Unlike some other connectivity optimization models (O'Hanley, 2011; Neeson et al. 2015), our model considers the importance of maintaining access to multiple types of habitat. Different riverine habitat types usually maintain different communities (Higgins et al., 2004; Erős, 2007). Diversification of habitat types within an RPA network can help to ensure the maximization of biodiversity (including community types). At regional scales, the common-sense approach (as we have done here) is to select habitats in proportion to their natural proportions within the landscape. This ensures that habitat complexity within the protected area network mirrors that of the wider landscape and that a natural pattern of biodiversity is maintained (Beechie et al., 2010; Thorp et al., 2010; Peipoch et al., 2015). Nevertheless, our model provides decision makers with full flexibility in terms of specifying the composition of an RPA network. For example, from the viewpoint of connectivity restoration for potamal fish species, there is usually a preference for protecting mid- to high-order streams (King et al., 2017). Conversely, with future climate change likely to exert the strongest influence on headwater streams (Isaak et al., 2010), it is conceivable that one would prefer to protect climatically threatened low order streams. Either of these scenarios could be easily accommodated for by our model (i.e., by adjusting the habitat fractions α_h and or the segment weights w_s).

Results from our case study of four Hungarian river catchments show that impressive increases in connectivity-weighted habitat can be achieved through targeted selection of barrier removals, corroborating the findings of other studies (Cote et al., 2009; Branco et al., 2014; Neeson et al., 2015). We also observed that the benefits of barrier removal strongly depend on RPA network size – for the same number of barrier removals, significantly larger gains in connectivity-weighted habitat are produced as the size of the RPA network increases. This is because with larger RPA networks, a much larger number of subcatchments can potentially be selected, thus providing greater leeway as to which subcatchments to protect and how to connect them up through barrier removal. Our results show that outcomes are markedly poorer if habitat conservation and connectivity restoration decisions are made separately. In the worst case, only 52% of maximum connectivity-weighted habitat is achieved using a two-stage approach where conservation decisions are made first, followed by barrier removal decisions. We also found that RPA land purchases provide substantially greater benefits compared to barrier removals. Using a single budget for RPA acquisition and barrier removals, RPA purchase always made up the bulk of spend, ranging from 73 to 93%.

We found that the allocation of conservation resources were sometimes very unevenly distributed among different catchments. For example, for the smallest sized RPA network comprising 10% of selectable river length, 95% is concentrated in Lake Balaton. Although focusing on one or few target areas may make sense from a resource efficiency standpoint, it can be cause for concern from a social equitability viewpoint (Halpern et al., 2013). To address this, additional constraints could easily be added to our model to ensure each catchments receives a certain minimum level of protection. Added justification for adopting a more balanced allocation of resources might be provided if further analysis showed that overall connectivity-weighted habitat only marginally decreased as a result of including these supplemental constraints.

Our case study was framed at the multi-catchment scale, as opposed to an individual catchment (Milt et al., 2017). Previous studies have shown that great efficiency is attained from planning at large spatial scales (Neeson et al., 2015). From a practical standpoint, however, it may be necessary to carry out planning on a catchment by catchment basis. For example, our results suggest that conservation and close-to-nature forest management might be the best land use functions in large parts of the Sajó Catchment, whereas agricultural land use might be better suited in most part of the Zagyva and Marcal Catchments and in the southern part of the Balaton Catchment. In the Sajó Catchment, forestry is already the main land use function in several subcatchments and consequently, outdoor tourism (e.g., hiking, recreational fishing) could be developed further in this region, while still conserving biodiversity (a win-win solution). In the other catchments, where agriculture is the main land use, managers should be able to easily identify those subcatchments that are the most valuable for conservation, and then subsequently use our framework in the land use selection process.

Our modelling approach provides a set of solutions for prioritizing river conservation and connectivity restoration actions based on pre-specified resources and design criteria. However, in a real-world planning situation, modelling and evaluation should be done in an iterative fashion, with active involvement of decision makers (Jax et al., 2013; Grizetti et al., 2016; McKay et al., 2017, Moody et al., 2017) in setting model parameters and performing what-if analyses. For example, as our case study showed, which subcatchments are selected can depend largely on the size of the RPA network and barrier removal budget. This suggests that land use planners and stakeholder groups (e.g., water authorities, national park authorities, fisheries groups) should ideally be involved in specifying the spatial extent of the analysis, determining realistic conservation targets / barrier removal budgets, and in evaluating how well conservation and ecosystem service needs are met. Their involvement would be particularly useful if more reliable data could be provided on land acquisition and barrier removal cost to help refine the analysis. Also, because outcomes will strongly depend on the set of ecosystem services (and indicators) used in the analyses (Nelson et al., 2009), involvement of planners and stakeholders groups in the earliest phases of the planning procedure is essential (Jax et al., 2013).

Finding a balance between conservation and ecosystem services provision is a complex and difficult task. There is no a single holy-grail solution that can be used to meet this need (Prager et al., 2012; Terrado et al., 2016). The modelling framework presented in this paper will invariably help conservation management to better account for connectivity restoration in conservation planning, resulting in more effective catchment scale maintenance of biological integrity and ecosystem services of riverscapes.

Authors' Contributions

TE, JO'H, and IC conceived and designed the study. IC and TE collected and analyzed primary research data; JO'H developed the optimization model and performed analyses of model results. TE and JO'H led writing of the manuscript. All authors contributed to editing manuscript drafts.

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Data Accessibility

Data available from the Dryad Digital Repository. DOI:

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Tables

Tab. 1. River habitat amounts, land use percentages, and number of artificial barriers in each river catchment. For river habitat, labels SMS, HLS, HLR, LLS, and LLR correspond, respectively, to submontane stream, highland stream, highland river, lowland stream, and lowland river. For land use, labels ART, AG, FOR, NFOR, WET, and WB correspond, respectively, to artificial surfaces, agriculture, forest, non-forest, wetland, and water bodies.

	Habitat Amount (km)						Land Use (%)						
Catchment	SMS	HLS	HLR	LLS	LLR	Total	ART	AG	FOR	NFOR	WET	WB	No. of Barriers
Balaton	0.0	321.1	49.3	189.0	37.8	597.2	6.1	44.6	27.0	5.6	2.7	13.9	138
Marcal	20.9	157.9	0.0	252.6	70.4	501.8	5.5	64.9	24.2	5.2	0.1	0.1	50
Sajó	103.7	424.8	294.0	63.0	0.0	885.5	7.2	53.4	31.3	7.7	0.3	0.1	52
Zagyva	25.7	267.4	0.0	322.8	67.3	683.3	6.6	66.2	21.1	5.5	0.3	0.3	75
All	150.3	1171.1	343.3	827.4	175.6	2667.7	6.4	56.4	25.8	6.0	1.0	4.4	315

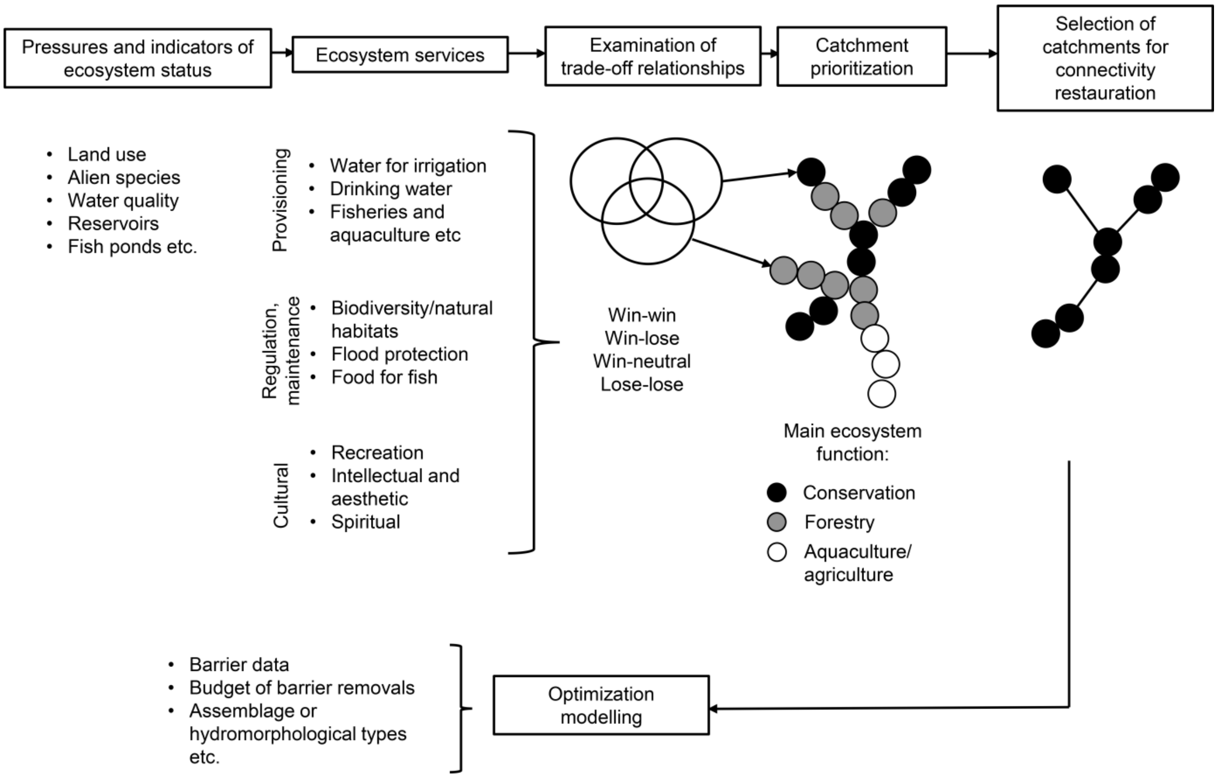


Fig. 1. A general framework for prioritizing catchments for biodiversity conservation versus ecosystem services and targeting connectivity restoration actions.

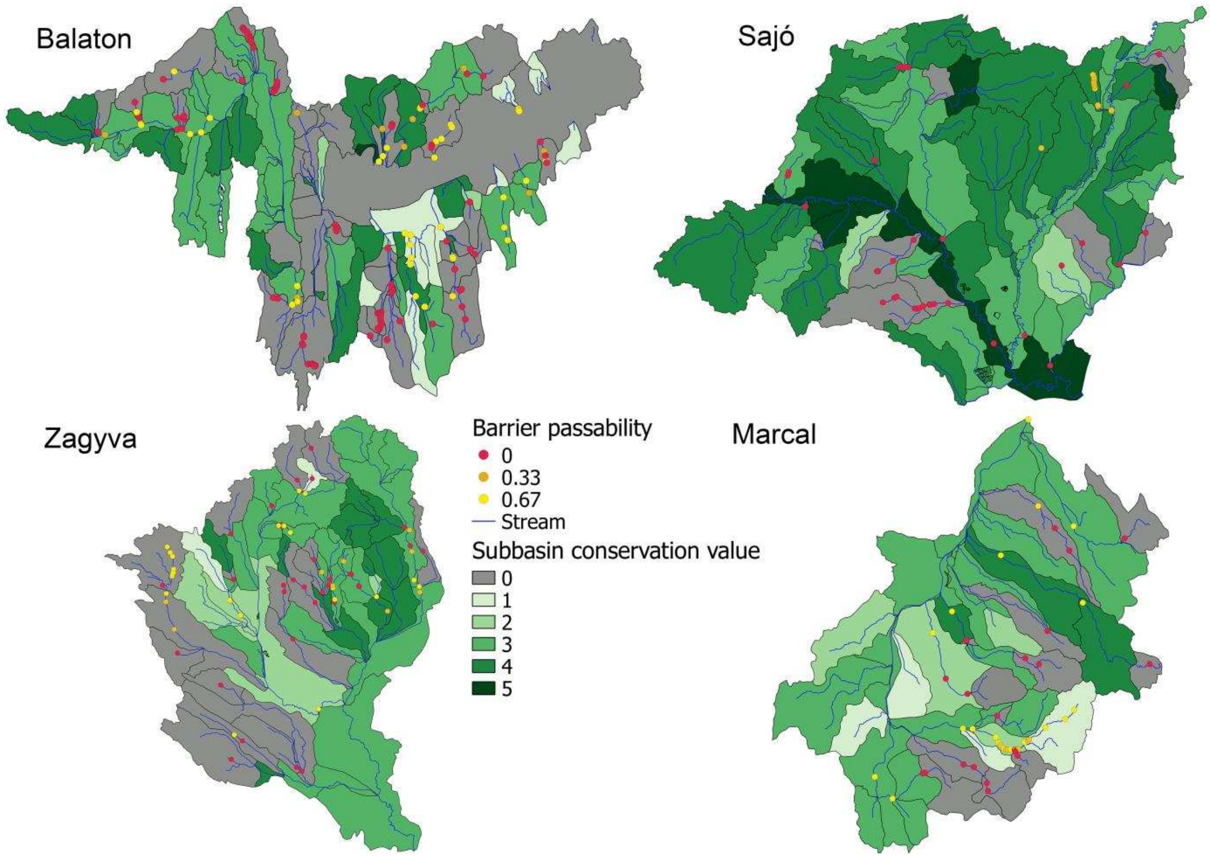


Fig. 2. Spatial pattern of biological integrity (BII) and distribution of artificial barriers in the four case study catchments: Lake Balaton, the Marcal River, the Sajó River, and the Zagyva River. BII is shown on a five-point scale, where a darker shade of green indicates higher integrity. Grey colored catchments have been assigned an integrity score of zero, indicating they were deemed better suited to land use functions other than conservation/connectivity restoration (e.g., agriculture). Note, that fully passable barriers (i.e. where barrier passability value equals 1) are not shown on the maps.

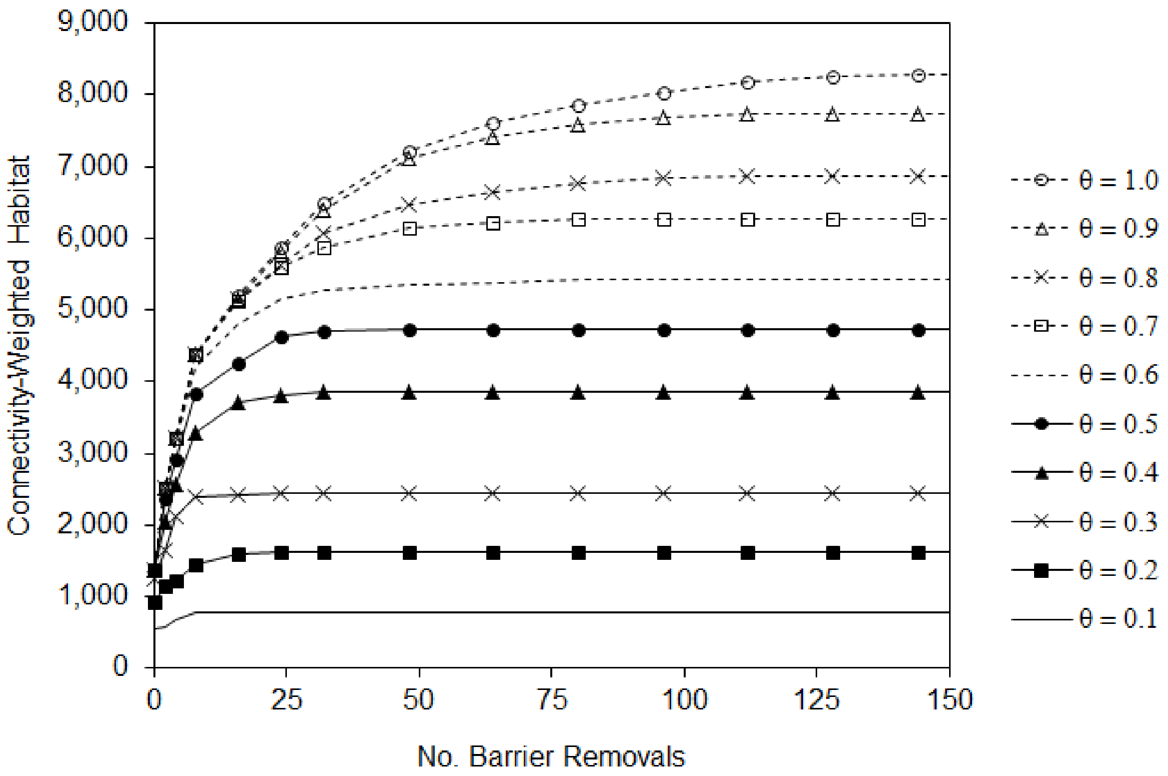
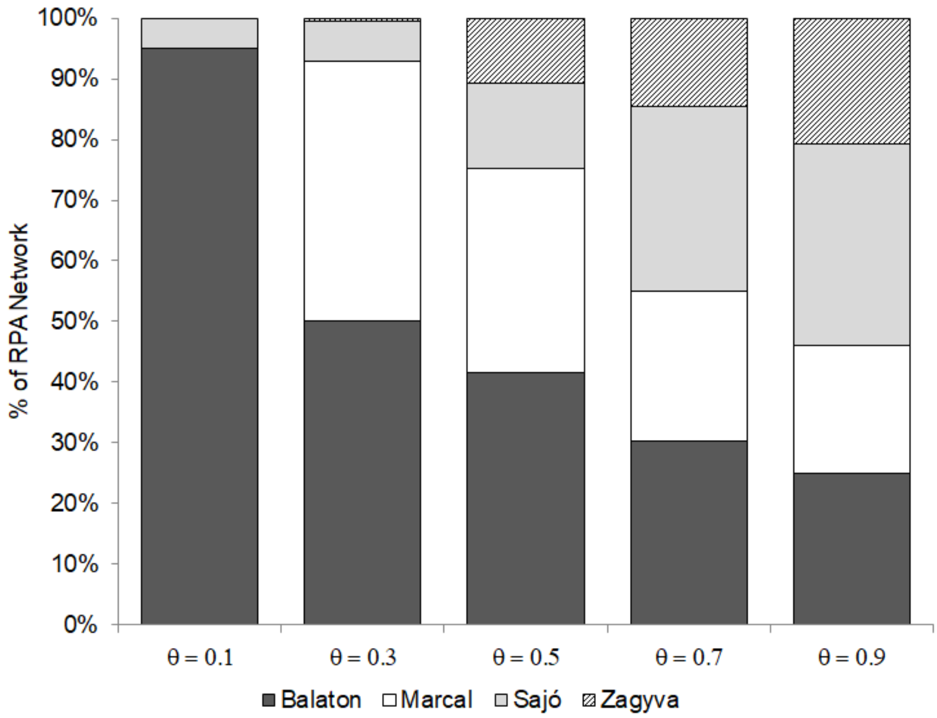


Fig. 3. Connectivity-weighted habitat versus number of barrier removals for various sized river protected area (RPA) networks.

(a)



(b)

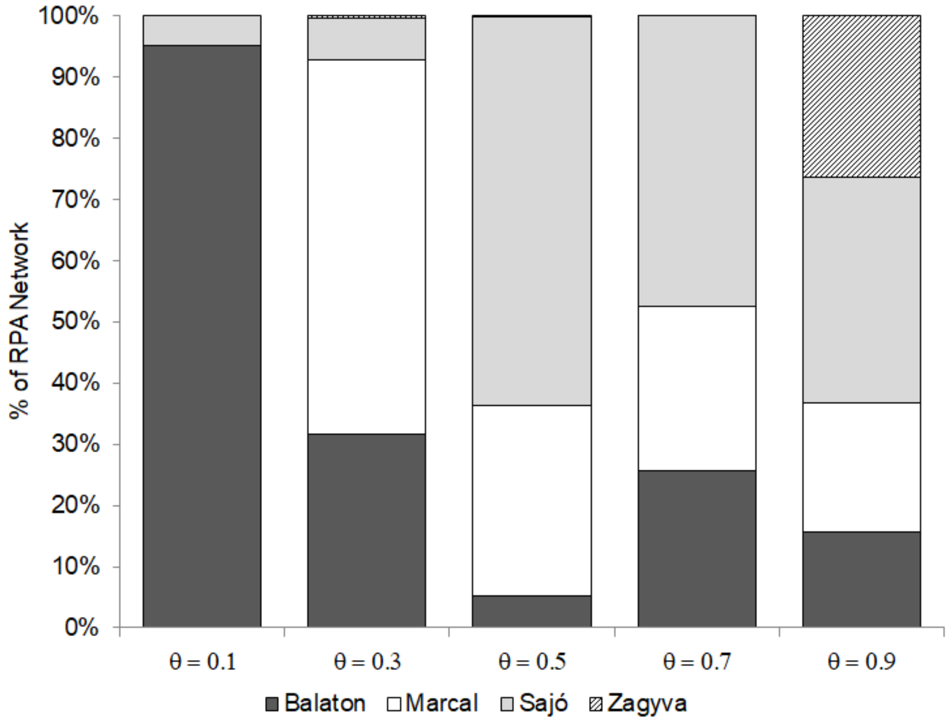


Fig. 4. Fraction of the RPA network in each river catchment given no barrier removal (a) and unlimited barrier removals (b) for various RPA network sizes.

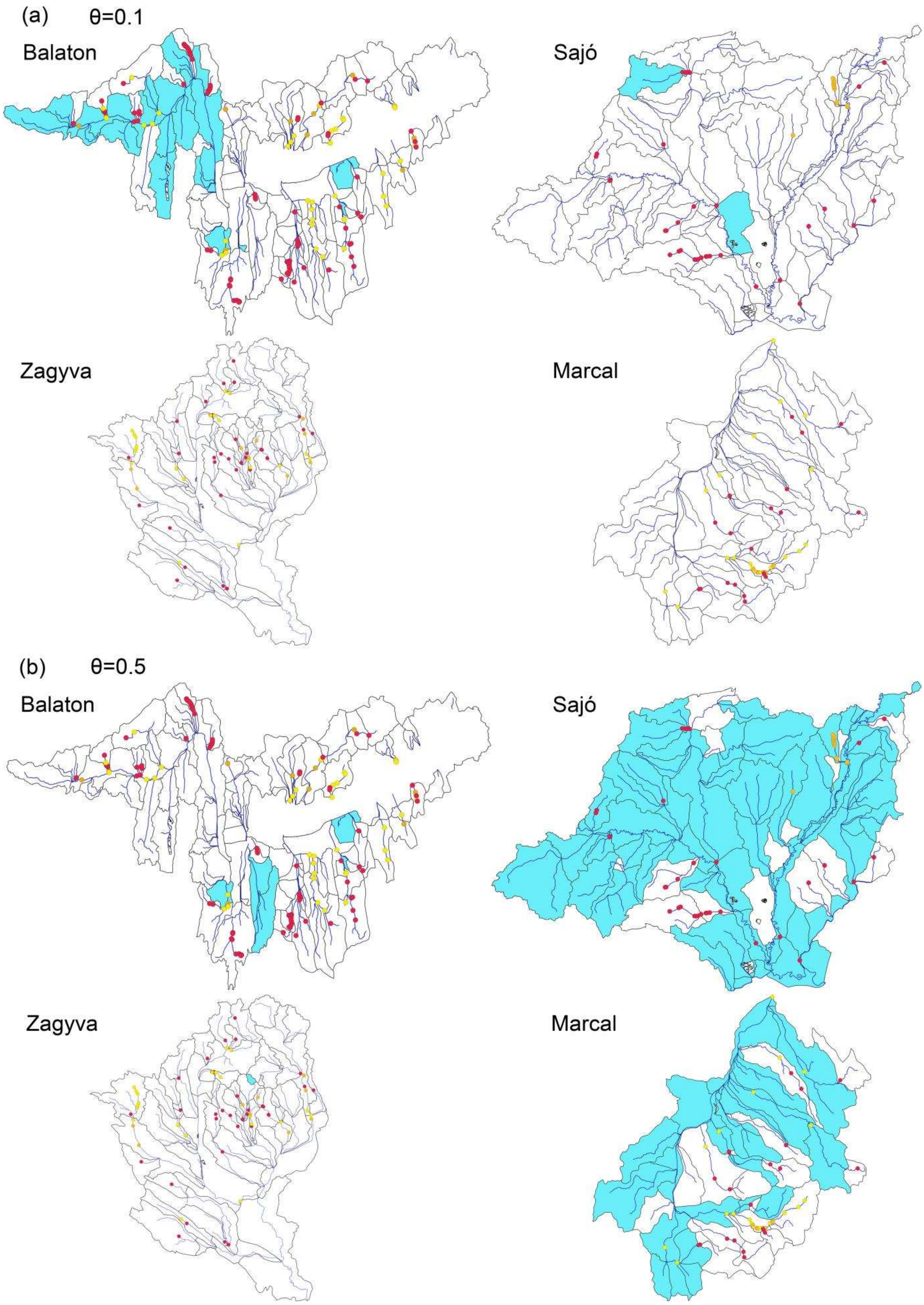


Fig. 5. Maps showing selected subcatchments for RPA networks of size $\theta = 0.1$ (a) and $\theta = 0.5$ (b) given unlimited barrier removals.

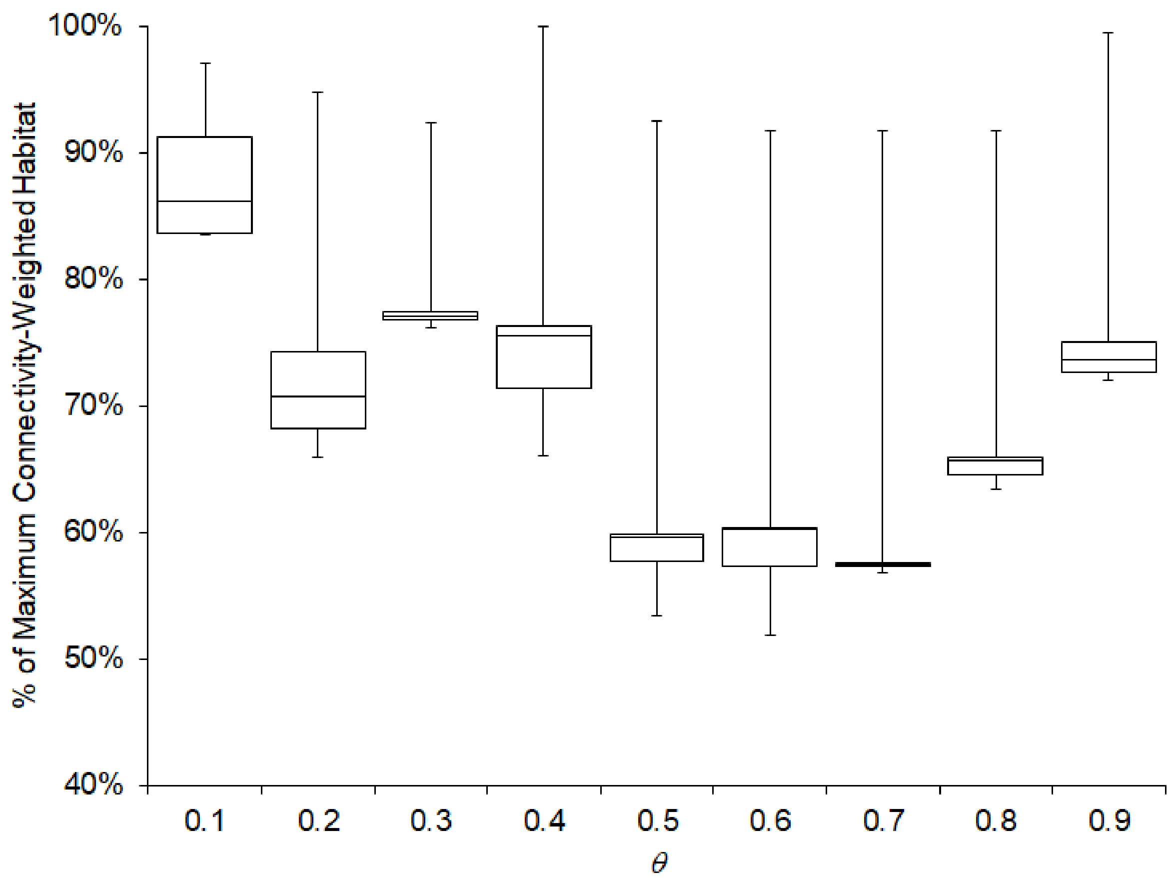
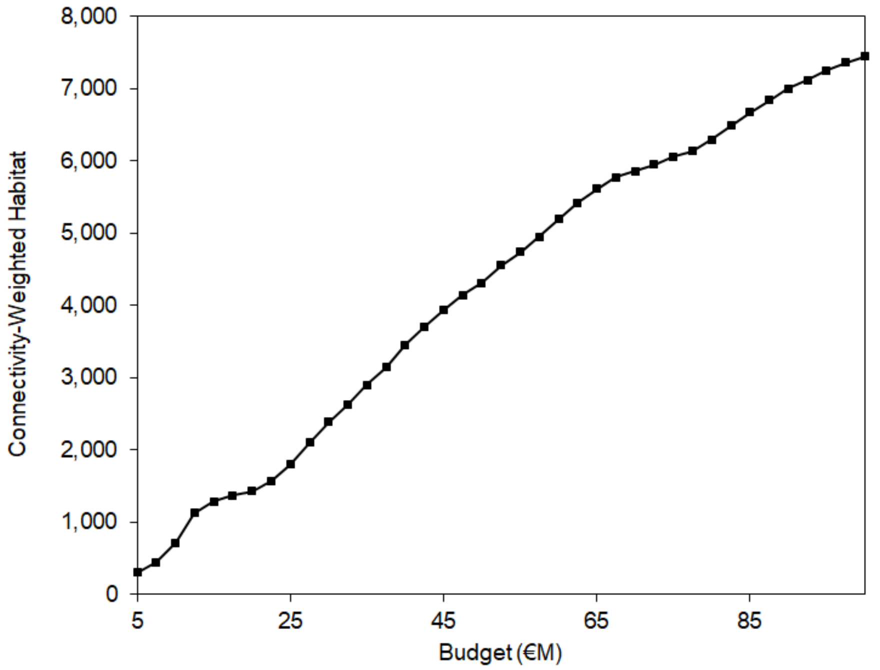


Fig. 6. Box plots showing the median, lower/upper quartiles, and minimum/maximum (whiskers) amount of connectivity-weighted habitat as a percentage of maximum for various RPA network sizes based on a sequential, two-stage approach to conservation and restoration planning (river protection decisions made first, barrier removal decisions second).

(a)



(b)

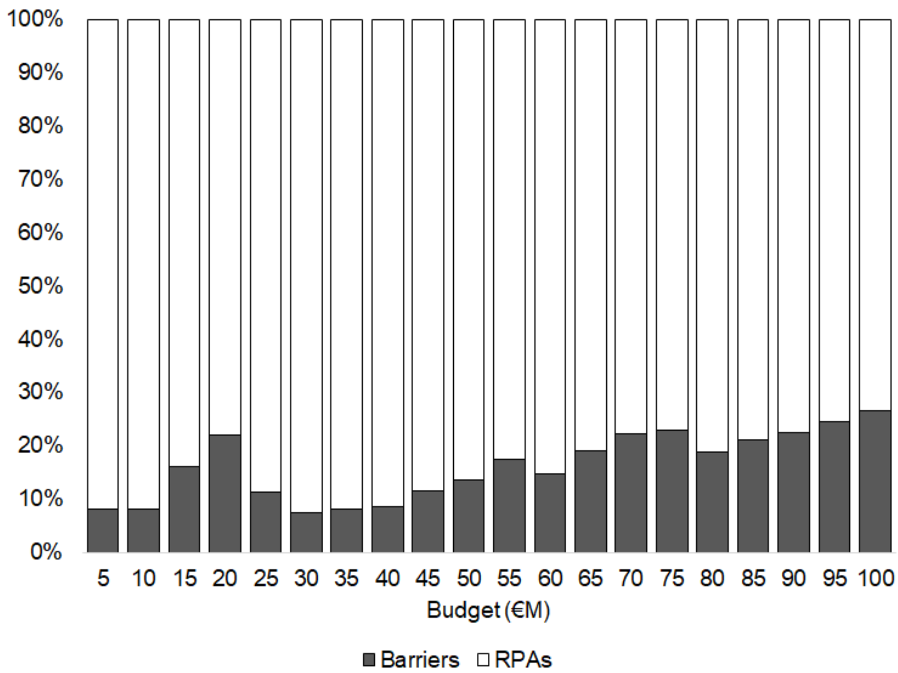


Fig. 7. Connectivity-weighted habitat versus combined budget for RPA acquisition and barrier removals (a) and relative spend on RPA acquisition versus barrier removal for various budget amounts (b).