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Commonalities in stream connectivity restoration alternatives: an attempt to simplify barrier removal optimization

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Abstract. Movement within stream corridors is a basic life history requirement of many aquatic organisms. Barrier removal in streams has become a common practice in the United States aimed to restore organism dispersal and meet conservation objectives; however, there are social and economic costs to the removal of barriers. Accordingly, tools to prioritize barrier removal, particularly optimization techniques, can be used to evaluate cost-benefit trade-offs. Many of these techniques, however, require programming experience and are not available to natural resource managers. Furthermore, conservation objectives vary considerably depending on the life histories of organisms under consideration, and these opposing objectives, in conjunction with variant socioeconomic costs, will influence optimization solutions, specifically which barriers to remove. To promote the use of optimization tools, straightforward and open-access platforms are needed to support use by managers, while also providing general approaches for holistic basin-scale connectivity restoration. Herein, we use two case studies, White Oak Creek (small watershed) and the Roanoke River Basin (large basin), to explore the divergent outcomes stemming from different conservation objectives and socioeconomic costs used to prioritize barrier removal. We conducted optimization modeling using a widely accessible platform along with an open-access solver plug-in to support a wide variety of conservation objectives. We used simple approaches to find commonalities in barriers identified for removal among divergent conservation objectives and provide alternative (i.e., hybrid removal-passage) strategies for approaching habitat restoration for diverse aquatic communities while increasing social benefits (i.e., hydropower energy). As expected, different conservation objectives aimed to support varied species life histories (e.g., diadromy, large-river vs. small-river potamodromy) have very different effects on optimization solutions. In both case studies, however, commonalities in solutions were identified through clustering groups of barriers into general connectivity restoration strategies. Furthermore, strategy types for a given barrier could be predicted with ≥72% accuracy using only four metrics. This suggests that optimization results can be simplified into general standards to support adoption of sustainable basin connectivity criteria strategies. Our framework provides a flexible and open-access approach to conduct relatively complex optimization modeling for stream barrier prioritization, while examining potential for agreement among divergence conservation objectives.

Key words: barriers; dam removal; fish passage; fragmentation; aquatic connectivity; landscape ecology; optimization; river networks; stream; watersheds.

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Introduction

Movement or change in spatial location through time (Nathan et al. 2008) is essential for many organisms to complete their life histories and ensure the persistence of subsequent generations. Preserving connectivity among habitat patches is vital to sustaining populations and communities by facilitating gene flow among populations (Bohonak 1999), colonization–extinction dynamics (Levins 1969), reproductive life histories (Taylor et al. 1993), and access to refugia (Sedell et al. 1990). Indeed, much of our understanding of the importance of habitat connectivity has stemmed from studies assessing impacts of habitat fragmentation (Laurance 2008), especially in stream environments. Barrier-imposed limitations have been shown to influence species distributions (Han et al. 2008, Reid et al. 2008), alter population dynamics (Fagan 2002), impede gene flow (Roberts et al. 2013), and limit ecological recovery following restoration (Brederveld et al. 2011).

Not surprisingly, the removal of barriers has become increasingly common (Bellmore et al. 2017). For example, over 1,000 dams have been removed across the United States (O'Connor et al. 2015*a*, American Rivers 2018), the majority of which have been in the last two decades (Bellmore et al. 2017). Additionally, culvert removal, replacement, or upgrades have become increasingly common (Favaro et al. 2014) and methods for addressing road crossing barriers have been widely adopted by major U.S. federal landholding agencies (Clarkin et al. 2005). Furthermore, the ecological benefits of dam or culvert removal activities have been clearly documented (Bednarek 2001, Kiffney et al. 2009, Hitt et al. 2012). Given the recent social impetus for barrier removal in streams, research on approaches and tools to prioritize barrier removal to support conservation objectives has increased in recent years (Zheng et al. 2009, O'Hanley 2011, Hermoso et al. 2017, Moody et al. 2017, Milt et al. 2018). Many of these approaches use optimization techniques to identify ideal barriers to remove to meet conservation objectives; however, optimization typically requires programming experience, which may be unavailable to natural resource managers. In response to this demand, decision support tools have become increasingly available, some of which provide dynamic optimization and visualization (Martin and Apse 2013, Moody et al. 2017).

Within mathematical optimization routines, the goals for optimization are reflected in objective functions and may consider one or more constraints (King et al. 2017). Objective functions are mathematical functions that seek to minimize or maximize numerical values based on a set of objectives, and even slight modifications to objective functions and constraints will influence results. In the case of barrier removal in streams, conservation objectives depend upon the life histories of organisms under consideration and habitat gain is typically maximized simultaneously with minimization of costs associated with barrier removal (McKay et al. 2017). Migratory behaviors, and the types of aquatic environments required to complete life cycles, vary among aquatic organisms; hence, objective functions and constraints will also vary and influence decisions on which barriers are most suitable for removal to address conservation objectives. For instance, diadromous fish migrate between ocean and freshwater environments to complete their life cycle whereas potamodromous fish migration occurs completely within freshwater. To enhance access to habitat for diadromous fishes, optimization routines would seek to remove barriers in order to maximize connectivity between river and ocean habitats (Kuby et al. 2005). Alternatively, in the case of potamodromous fish, connectivity among river patches would be maximized without consideration of proximity to the ocean (O'Hanley et al. 2013, King et al. 2017). In basins where multiple aquatic organisms are of conservation concern, objective functions aimed to inform barrier removal for one group of species are unlikely to be representative of habitat needs of other species. However, barrier removals come at a cost, both monetarily and, in some cases, as loss of services provided by dams and other structures (Fausch et al. 2009, Erõs et al. 2018). In addition, ecological trade-offs exist between less- and more-connected river networks (Fausch et al. 2009, Jackson and Pringle 2010).

Studies using optimization approaches to prioritize barrier removal have primarily focused on a single conservation objective, with most cases aimed at restoring habitats for diadromous

fishes (Kuby et al. 2005, Kocovsky et al. 2009), or potamodromous fishes (O'Hanley et al. 2013, Ziv et al. 2012), but rarely both. Considering both life histories, however, may still not address habitat needs of all species of concern (Kemp and O'Hanley 2010). Objective functions commonly include maximizing habitat area or stream length, whereas in a few cases, indices, such as richness or biomass, are used (Ziv et al. 2012, King et al. 2017). Additionally, some studies have included the cost of barrier removal in multi-objective optimization (King et al. 2017, Moody et al. 2017) whereas others have optimized purely on the grounds of ecological benefit (Kocovsky et al. 2009). Similar to variant conservation objectives, there are many approaches to evaluate or quantify the cost of barrier removal. Moreover, opposing conservation objectives become increasingly difficult to reconcile in cases of limited resources to support those objectives. Ultimately, approaches to find convergence among numerous opposing conservation objectives are needed to guide habitat restoration for diverse aquatic communities. Furthermore, for managers to readily use decision support tools, there is a need to simplify optimization solutions into general criteria that inform holistic ecological sustainability criteria (Jager et al. 2015).

Herein we evaluate how differing conservation objectives aimed to improve habitat access influence the prioritization of barriers for removal. Our goal was to examine whether the potential exists for consistency in the selection of barriers for removal when conservation objectives targeting fishes with unique life histories were compared. To accomplish this goal, we used two case studies, a small watershed and a large basin, to represent different geographic contexts, network configurations, life histories of species of conservation importance, and societal costs of dam removal. In both cases, we aimed to create scenarios representative of divergent life history endpoints in order to determine the possibility for addressing holistic ecological sustainability criteria (proposed by Jager et al. 2015) while reconciling opposing stream connective restoration alternatives. Because barrier removal does not come without a cost, we optimize cost-benefit trade-offs (i.e., optimization modeling) by considering two forms: the monetary cost of physical removal (based on the size of the barrier) and the potential loss of hydroelectric energy generation, either actual energy loss (in the case of hydroelectric dams that are removed) or potential energy loss (in the case of non-hydroelectric dams removed that could otherwise produce electricity if retrofitted with generators).

Promoting widespread implementation of barrier prioritization requires putting the technology and ability in the hands of those that are most likely to utilize it (McKay et al. 2017). Our work addresses this challenge by conducting optimization modeling using a platform to which most managers already have access and familiarity (i.e., Microsoft Excel). The platform is modular in that managers can adapt the framework for use in their specific conservation applications. For both case studies, we provide relatively straightforward approaches for seeking commonalities among restoration alternatives or proposing hybrid approaches to restoration. Finally, we determine if there are simple metrics for determining likely optimal solutions in cases where there are insufficient resources for optimization modeling.

Methods

The two case studies include White Oak Creek, a small watershed in Tennessee, USA, and the Roanoke River Basin, a large river system extending across Virginia and North Carolina, USA. White Oak Creek provides a perspective of the optimization of road culvert and weir removal to benefit life histories of fish species commonly found within 1st- to 4th-order stream systems. In small watersheds, costs primarily include the monetary considerations of removing the barrier or replacing road culverts with ecologically friendly designs (Moody et al. 2017). In contrast, the Roanoke River Basin provides an ideal comparison of a basin-scale context where large barriers, primarily dams, are considered for removal to support the conservation of fish and mussel communities occupying a range of river systems from headwaters to estuaries. In this case, cost of barrier removal could be monetary, but also the potential loss of energy production from hydropower. That is, dam hydropower can be quantified as a cost associated with removal, including potential energy capacity (measured in megawatts; MW) and potential

energy generation (measured in megawatt hours; MWh) lost if a dam is removed.

White Oak Creek, Tennessee

White Oak Creek (WOC) is a small (17 km^2) and highly fragmented watershed located within the U.S. Department of Energy Oak Ridge Reservation near Oak Ridge, Tennessee, USA (Fig. 1). The creek runs through the main campus of Oak Ridge National Laboratory and eventually becomes a 4th-order system at its terminus with the Clinch River. The topography and hydrology of WOC are characteristic of streams in the Ridge and Valley Ecoregion with elevations ranging from 226 to 413 m and headwater portions of the catchments experiencing drying during low-flow months. Most of the watershed is forested; however, the mainstem portion of WOC that runs through the ORNL campus is channelized and receives treated and untreated effluents from facility discharges that are sources of contaminants and altered temperature (McManamay et al. 2016). The degree of fragmentation in WOC is relatively high (1.85 barriers/km^2) in comparison with that reported for other watersheds (0.03–0.4; Meixler et al. 2009, Favaro et al. 2014). The watershed contains 31 barriers (dams, weirs, or culverts), only 6 of which are fully passable to species occurring in the watershed (McManamay et al. 2016; Fig. 1); thus, we considered 25 barriers for removal (dams, weirs, flumes) or replacement (culverts). Two major dams near the mouth of WOC prevent migration of fish species in or out of the Clinch River system. WOC currently has 22 native species and 5 invasive or non-endemic species (Appendix S1). Most common native species in WOC include Blacknose dace (*Rhinichthys atratulus*), Largescale stoneroller (*Campostoma oligolepis*), and Creek chub (*Semotilus atromaculatus*).

Scenarios and data.—Three scenarios were selected to illustrate different conservation

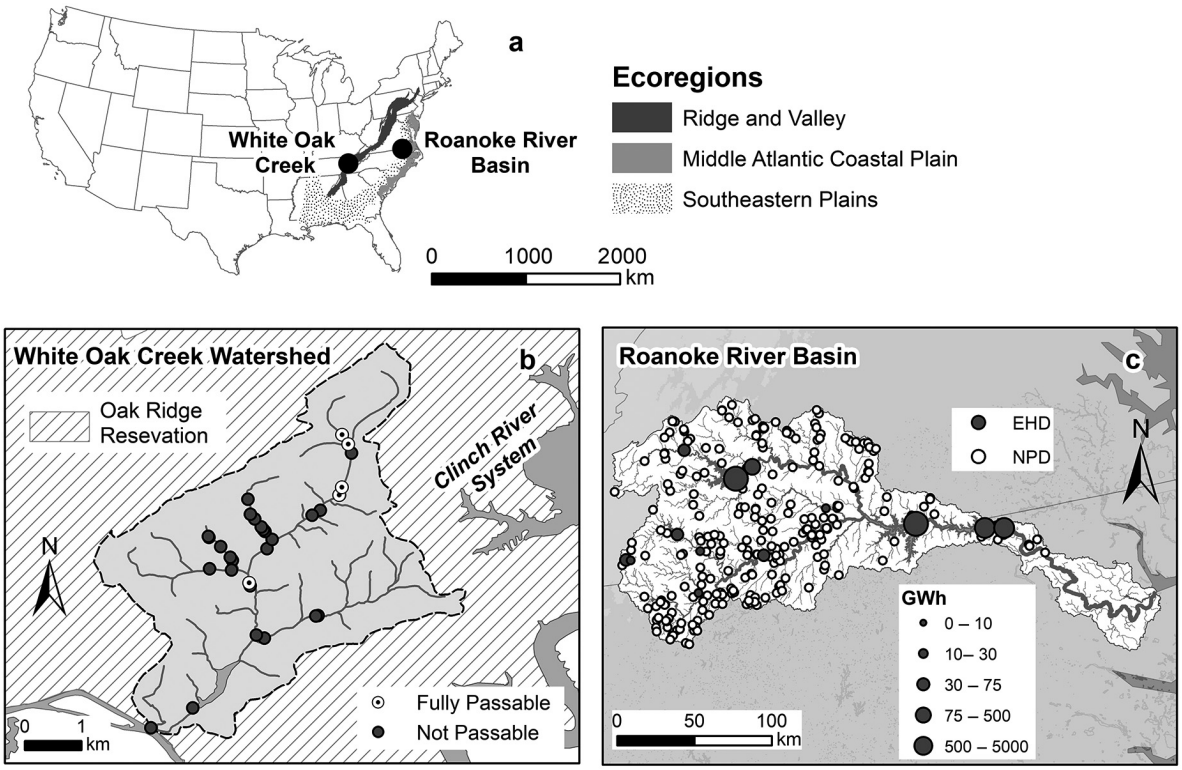

Fig. 1. Study site map of (a) locations of White Oak Creek (WOC) watershed and the Roanoke River Basin (RRB) within different ecoregions, (b) inset of WOC and 31 barriers mapped according to their passage potential, and (c) inset of the RRB depicting existing generation from hydropower dams (EHDs) and potential generation from non-powered dams (NPDs).

objectives, representing the variant life histories of stream fish detected or with prospect of reintroduction or recolonization in WOC. Because WOC is a small tributary of the Clinch River system (an important source of colonists and larger migratory fish), it provides a scaled-down representation of a larger basin entering an estuary and is analogous to conservation efforts promoting river access for diadromous fishes. We term the first scenario pseudo-diadromous as it prioritizes the removal of barriers in a downstream-to-upstream direction to maximize seasonal migration of freshwater fish from the Clinch River at the base of the network (consistent with diadromy) but fishes only move between the Clinch River and WOC; they do not access salt water (inconsistent with diadromy). Objectives of the second scenario optimize habitat to support the migratory behavior of potamodromous fish completely within the WOC watershed or between WOC and the Clinch River. In this case, the directionality of barrier removal within the river network is unconstrained. The third scenario prioritizes barrier removal to support conservation of Tennessee dace (*Chrosomus tennesseensis*), a state-listed species of concern in Tennessee. Tennessee dace is a small fish that migrates during spring from mainstem creeks into small headwater streams when flow conditions are conducive to spawning. They occur in adjacent watersheds but not WOC and are a candidate for reintroduction. In contrast to the pseudo-diadromous scenario, barriers are prioritized in an opposite fashion and are selected to maximize connectivity between headwaters and mainstem creeks rather than connectivity between mainstem and downstream sources.

In all three scenarios, stream patch connectivity is measured as the distance of stream network segments made accessible to other stream segments following the removal of barriers or replacement of impassable culverts with a fish-friendly design. We used the National Hydrography Dataset (1:24k) to represent the stream network in WOC. Network Analyst (ArcMap 10.3, ESRI, Redlands, California, USA) was used to measure the unobstructed river distance upstream (i.e., functional distance; Cote et al. 2009) of each barrier that would be accessible if that barrier was removed or replaced. Costs (USD) were determined separately for culvert replacement and for removal of weir, flumes, or dams. Culvert replacement included removal of the existing closed pipe structures and installation of a more fish-friendly corrugated metal pipe arch culvert. Arch culverts leave the streambed uncovered and have been shown to be superior in passing fish than traditional pipe culverts (Gibson et al. 2005). Estimated costs of replacement are described in Appendix S2 and include materials, labor, and equipment required for road removal, excavation, removal of the existing structure, erosion control, installation of new structures, and road construction (among several other items). Several culvert replacement scenarios of estimated costs per length were developed depending on requirements for the dimensions of the structure and stream channel (Appendix S2: Table S1).

Weir and dam removal costs were compiled from multiple sources, if dam height (h) was also included (in m; Appendix S2: Fig. S1). Because dam removal costs (C_R) exponentially increase and then reach an asymptote with increasing dam size, we developed a log–log relationship ($r^2 = 0.68$) between cost and dam height (1).

$$\log(C) = 2.29 * \log(h + 1) + 3.41 \qquad (1)$$

Flume removal costs were determined based on both structure height (using Eq. 1) and the length of concrete pads. Presumably, removal of concrete pads would require channel remediation and bank stabilization, which, on average, costs \$237 m^{-1} of stream (Appendix S2: Table S2). The total cost in 2017 USD was the sum of the two costs applied to each structure.

Objective function.—We prioritize barrier removal using a multi-objective benefit-vs.-cost function modified from Kuby et al. (2005). All scenarios for WOC maximize the connectivity of stream patches while minimizing the cumulative cost of barrier removal or replacement (RR):

$$\frac{\sum_{i=0}^{j} D_i X_i}{\sum_{i=0}^{n} D_i} * w - (1 - w) * \frac{\sum_{i=0}^{j} C_i X_i}{\sum_{i=0}^{n} C_i} \qquad (2)$$

Subject to:

$$X_i \in \{0, 1\} \quad \text{for all } i \qquad (3)$$

$$\sum_{i=0}^{j} C_i X_i \leq T \qquad (4)$$

and one of the following:

$$X_i \leq X_j \text{ for all } i \text{ and } j, \text{such that } j \text{ is directly downstream of } i, \text{or} \quad (5)$$

$$D_i \geq 500\,\text{m} \text{ or } X_i \leq X_j \text{ for all } i \text{ and } j, \text{such that } j \text{ is directly downstream of } i, \text{ or} \quad (6)$$

$$X_i \geq X_j \text{ for all } i \text{and } j, \text{such that } j \text{ is directly downstream of } i \quad (7)$$

where D is the unobstructed river distance upstream of barrier i (i.e., the functional distance) and C is the cost of RR for barrier i. X is a binary selection variable for each barrier, where $X_i = 1$ if barrier i is selected for RR, otherwise $X_i = 0$. For a given solution and set of constraints, there are j barriers selected for RR, out of $n = 25$ total barriers. Hence, the denominators represent cumulative stream network distance (32.8 km) and cumulative costs of RR ($2.4 M USD) for the entire watershed and are used to standardize distance and costs from 0 to 1. w is an optional weighting factor, ranging from 0 to 1, that provides emphasis on distance vs cost in the objective function. T is the total cumulative cost threshold allowed at any time.

Scenarios are differentiated from each other using varying constraints. For constraint (5), the pseudo-diadromous scenario, a given barrier can only be subject to RR if and only if the barrier immediately downstream has been selected for RR. In constraint (6), the potamodromous scenario, barrier i was available for RR if either functional distance exceeded 500 m or similar conditions for (5) were met. The constraint for the Tennessee dace scenario (7) required that we reverse the constraint for the diadromous scenario. Because any barrier may have more than one barrier immediately upstream (i.e., mainstem and tributaries), constraint (7) was flexible in that a given barrier could be subject to RR if any of the immediate upstream barriers were first selected for RR.

For each of the scenarios above, we optimized objective (2) under varying incremental increases in T to develop trade-offs curves of distance opened vs. total cost. We explored 20 values of T each under three different values of w: 0.5, 0.7, and 0.9, to examine the influence of weighting distance more than cost (60 total solutions for each of the three scenarios). This differed from Kuby et al.'s approach where trade-off curves were obtained by evaluating the objective function over many different values of w. All scenarios were constructed on spreadsheets within Microsoft Excel as it provides very flexible functionality in building interdependent constraints based upon the spatial arrangement of barriers, such as the conditionality of an upstream barrier being available for removal contingent upon the removal of a downstream barrier (i.e., pseudo-diadromous scenario). Hence, with each iteration or trial solution, the conditions of barriers dynamically change in relation to the decision status of other barriers. Spatial dependencies among barriers were constructed by specifying topological (i.e., upstream-to-downstream) connections among barriers. Dynamic changes in spatial relationships arising from barrier RR were modeled using Excel's VLOOKUP function.

To solve these solutions, we used OpenSolver, an open-access Excel Visual Basic for Applications (VBA) add-in (Mason 2012), which employs the Computational Infrastructure for Operations Research Branch and Cut algorithm (COIN-OR 2016), a mixed integer linear programming solver written in C++ (Lougee-Heimer 2003, Forrest and Lougee-Heimer 2005). OpenSolver uses a parser that directly translates VBA language from Excel spreadsheets into C++; thus, it is robust to discontinuous constraints imposed by VBA search algorithms (e.g., VLOOKUP function). Most solutions were obtained in ≤1s.

Finding convergence.—Because there were only 25 barriers, we simply examined the total frequency that each barrier was selected under all three scenarios. Barriers selected by individual scenarios and combinations of scenarios were compared. Barriers selected under all three scenarios were prioritized as convergent solutions.

Roanoke River Basin, North Carolina and Virginia

The Roanoke River originates in the Blue Ridge Ecoregion of southwest Virginia and traverses the Piedmont, Southeastern Plains, and Middle Atlantic Coastal Plain Ecoregions before draining into the Albemarle Sound in North Carolina (Fig. 1). At its terminus, the river drains a 25,000-km^2 area and average annual discharge is 260 m^3/s. The river's largest tributary, the Dan River, drains the southwestern portion of the basin and enters the Roanoke River at

approximately one-third of its course to the ocean. Over 93 fish species and 14 mussel species inhabit the basin (Appendix S3). Of these, eight fish and nine mussels are either critically imperiled, imperiled, or vulnerable according to the NatureServe Global Conservation Status (Faber-Langendoen et al. 2012), whereas many more species are considered species of concern at the state level. Three fish and one mussel species are federally endangered, and two fish species are federal species of concern (Appendix S3). At least nine species of diadromous fish inhabit the lower Roanoke River basin, including Atlantic sturgeon (*Acipenser oxyrinchus*) and American eel (*Anguilla rostrata*).

The Roanoke River Basin (RRB) is highly fragmented by dams and reservoirs, the largest of which provide a considerable source of hydropower energy (Johnson et al. 2013). Within the basin, 13 existing hydropower dams (EHDs) have a cumulative capacity of 1200 MW and generate 6140 GWh per year. Non-powered dams (NPDs) are far more numerous (245), but the potential energy capacity and generation from these facilities, if developed, would constitute a small fraction of that provided by existing hydropower dams (9.4 MW and 23 GWh, respectively; Johnson et al. 2013; Fig. 1). Roanoke Rapids Dam is the downstream-most barrier on the mainstem Roanoke River, after which the river flows freely for 326 km before entering the estuary.

Scenarios and data.—We used four scenarios representing divergent conservation objectives in the RRB; two of these were analogous to those developed for WOC except for some notable changes to the objective function and constraints. In contrast to WOC, we only evaluated the removal of dams to enhance migratory habitat and did not consider alternatives, such as fish passage (although we compare the cost of removal to that of fish passage). Additionally, we used the amount of hydropower energy lost to represent the cost of removing a given dam in objective functions for the RRB. In calculating the cumulative amount of energy, we included both the annual generation from existing hydropower facilities and the potential annual generation from non-powered dams, if they were developed. In contrast to WOC, generation was included in the objective function as a cost of barrier removal; however, monetary cost was included as a constraint similar to (4). Similar to WOC, the diadromous scenario for the RRB aimed to maximize migratory habitat corridors from the estuary to upstream tributaries, whereas the potamodromous scenario prioritized barrier removal based purely on thresholds of stream patch size (i.e., distance) without respect to direction. For the RRB, barriers were candidates for removal if functional networks were ≥20 km, as the occurrence of migratory riverine fish (e.g., *Moxostoma spp.*) tends to decline substantially in fragment lengths less than this value (Reid et al. 2008, McManamay et al. 2013). The third and fourth scenarios were different in that the selection of dams for removal was constrained by the occurrence of a fish or mussel species of concern within corridors that would be opened following barrier removal. In these cases, a barrier was a candidate for removal if the following conditions are met: (1) A species of concern occurs within the barrier's upstream functional network, (2) the barrier's functional network ≥20 km and a species of concern is found within any free-flowing reaches downstream, or (3) the barrier's functional network ≥20 km and a species of concern is found below the nearest downstream dam, which has been selected for removal. All but the fourth scenario used distance as a measure of ecological benefit in the objective function. For the fourth scenario, we used the number of fish or mussel species of concern that could benefit from removal.

We used the NHDplus V2 (1:100k) stream reaches to represent the stream network within the RRB. Locations of existing hydropower dams and non-powered dams, their energy capacity, and their current or potential energy generation were obtained through Oak Ridge National Laboratory's National Hydropower Asset Assessment Program (Samu et al. 2018). Resource assessments for the potential energy capacity (MW) and potential energy generation (MWh) from non-powered dams were developed by Hadjerioua et al. (2012). Functional networks upstream of each barrier were calculated similar to WOC using Network Analyst (ArcMap 10.3). Although we only included the cost of removal (1) in our objective function, we calculated the initial capital cost (ICC) in 2017 USD of constructing a fish passage facility (ICC_f) (Eq. 8; Hall et al. 2003) and powering a non-power dam

(ICC_p; Eq. 9, O'Connor et al. 2015*b*) for comparison to removal costs

$$ICC_f = 1.3 \times 10^6 * p^{0.056} \quad (8)$$

$$ICC_p = 10^7 * p^{0.954} * h^{-0.252} \quad (9)$$

where p is plant nameplate capacity (MW) and h is dam height (m).

Distributions of fish and mussel species in the RRB were compiled from multiple sources including the Global Biodiversity Information Facility (http://www.gbif.org/) and State Natural Heritage Program elemental occurrences (Virginia Department of Conservation and Recreation, Natural Heritage, http://www.dcr.virginia.gov/natural-heritage/; North Carolina Natural Heritage Program, http://www.ncnhp.org/) survey programs.

Objective function.—Similar to WOC, the objective functions for the RRB also represent a benefit-vs.-cost function, except that stream connectivity is maximized against cumulative electricity generation lost with the removal of one or a combination of barriers. For the first three scenarios, the function is the same as (2), except that cost is represented by electricity generation (MWh) lost with the removal of barrier i. Additionally, the constraints are similar to (3) and (4) and the denominators of function (2) represent the total length of stream (20,163 km) and total energy generation (6159 GWh) in the entire basin and are used to standardize benefit and cost terms from 0 to 1.

Constraints for the diadromous scenario in the RRB were the same as those of the pseudo-diadromous scenario in WOC. Additionally, the constraints of a given barrier being available for removal within the Roanoke potamodromous scenario were similar to those of WOC with some slight changes:

$$D_i \geq 20\,\text{km or } X_i \leq X_j \text{ for all } i \text{ and } j, \text{ such that } j \text{ is directly downstream of } i. \quad (10)$$

The third scenario, Species of Concern A (SOC_A), relies on function (2) for the objective, but requires that the following constraints be met:

$$B_i \in \{0,1\} \text{ and } X_i = X_i B_i \text{ for all } i \quad (11)$$

where B is a binary indication of whether the removal of barrier i would benefit habitat conditions for a species of concern. This requires considering the occurrence of species of concern within the functional network of barrier i, but also the functional network of barriers adjacent to barrier i and whether they have been selected for removal.

The benefit indicator $B_i = 1$ if at least one of the following constraints is met:

1. A species of concern is found in the functional network upstream of barrier i, or
2. $D_i \geq 20$ km, and
3. Barrier j is directly downstream of barrier i and $B_j = 1$, or
4. Barrier k is directly downstream of barrier j, which is downstream of barrier i, $X_j = 1$, and $B_k = 1$.

The fourth scenario, Species of Concern B (SOC_B), is similar to the previous scenario except that we maximize the cumulative number of species of concern benefiting from barrier removal relative to electricity generation lost, depicted as:

$$\frac{\sum_{k=0}^{m} S_k}{\sum_{k=0}^{33} S_k} - \frac{\sum_{i=0}^{j} E_i X_i}{\sum_{i=0}^{n} E_i} \quad (12)$$

Subject to:

Constraints (3) and (4), and

$$\text{if} \sum_{i=0}^{j} B_{ik} > 0, \text{ then } S_k = 1, \text{ else } S_k = 0, \text{ and} \quad (13)$$

$$B_i \in \{0,1\},\ S_k \in \{0,1\} \text{ for all } i \text{ and } k \quad (14)$$

where S represents whether species of concern k could benefit from the removal of barrier i based on the spatial occurrence of the species. For a given solution, m species could benefit from the removal of j number of barriers, out of a total of 33 species of concern. As in the previous scenario, B_{ik} is a binary indication of whether the removal of barrier i would benefit habitat for species of concern k. Species of concern k may occur in many locations and, hence, could benefit from the removal of any one of multiple different barriers. Thus, we impose the constraints above to ensure that species k is not counted more than once, and the maximum number of species benefitted cannot exceed 33. In order for B_i to equal 1, the same constraints are imposed here as in the SOC_A scenario.

We used 30 incremental increases in T (4) to develop trade-off curves of distance opened vs. energy lost. Because the complexity of our scenarios for the RRB was sufficient to provide examples of divergent objectives, we did not explore the effects of varying values of w. Similar to WOC, we constructed all scenarios using Microsoft Excel and solved solutions using OpenSolver.

Finding convergence.—Our analysis evaluated 30 incremental values of T for each of the four scenarios, resulting in 120 different solutions for 259 different dams. Hence, approaches to simplifying these solutions by examining patterns of convergence are warranted to inform management. We used a series of alternative Gaussian mixture models to cluster dams into groups based on the proportion of times each barrier was selected for removal under each scenario. Models varied according to the number of components (i.e., clusters), covariance structure, and shape/volume of ellipses. Models ranging from 2 to 20 clusters were fit using expectation-maximization parameter estimation and evaluated with Bayesian Information Criterion (BIC) using the mclust package in the R programming environment (Scrucca et al. 2016). After determining the model with the largest BIC value, we summarized multiple characteristics, such as the total monetary and energy costs of removal, for each cluster. We hypothesized that cluster membership could be predicted for each barrier using a few simple variables: D_i, stream distance to estuary, MWh, and the number of species of concern within the functional network above a barrier. We used recursive partitioning in the rpart package of the R programming environment (Therneau and Atkinson 2018) to develop classification trees for predicting cluster membership for all clusters and then a separate model for differentiating convergent solutions (i.e., more than 1 life history) from singular life history solutions.

As an alternative to simple clustering, we adopted a hybrid approach that simultaneously considered removal of some barriers, provision of fish passage at others, and potential increases in hydropower generation capacity at EHDs. Based on the previous analysis, we identified clusters with the highest frequency of barriers selected for removal. From these barriers, we identified fish passage as a restoration option at a subset of EHDs whereas removal was determined for NPDs. Upgrades to existing generation units (i.e., generator rewinds) are estimated to conservatively provide a 10% increase in generation at EHDs (DOE 2016). The initial 2017 capital cost (USD) of these capacity additions (ICC_c; O'Connor et al. 2015*b*) is estimated as:

$$ICC_c = 3.29 \times 10^5 * p^{0.753} \tag{15}$$

It should be noted that generator replacement costs are >10 times the ICC_c value (O'Connor et al. 2015*b*). We examine the cumulative benefits and costs (monetary and generation) of the alternatives.

Results

White Oak Creek, Tennessee

Trade-off curves and the selection of barriers under various cost thresholds varied among different scenarios (Fig. 2). The pseudo-diadromous scenario reached an asymptote of diminishing ecological benefit (i.e., 25 km of stream distanced opened) beyond costs of $800,000 whereas optimal ecological benefits (25 km) were observed at $500,000 for the potamodromous scenario. For the Tennessee dace scenario, increases in ecological benefits continued until cumulative expenditures reached $900,000, at which point almost 30 km of stream were opened. Varying weighting for distance vs. cost had little influence on barrier selection and trade-off curves. Scenarios also differed in the portions of the WOC network that would be opened following barrier RR (Fig. 2). Barriers prioritized for RR within the pseudo-diadromous scenario opened stream networks in the downstream portion of WOC whereas stream networks opened within the potamodromous scenario were centralized to the middle portion of WOC. Networks opened by barrier RR within the Tennessee dace scenario were similar to those opened by the potamodromous scenario except that barriers in small tributaries were also prioritized.

Ten of the 25 barriers were selected for RR under all three scenarios (Fig. 3). The stream network opened from these 11 barriers included a path from the mouth to the uppermost mainstem tributary (Fig. 3). Eight of the 11 barriers were selected at least 50% of the time across all scenarios and all cost thresholds whereas the remaining 3 were selected at least 30% of the time. Five

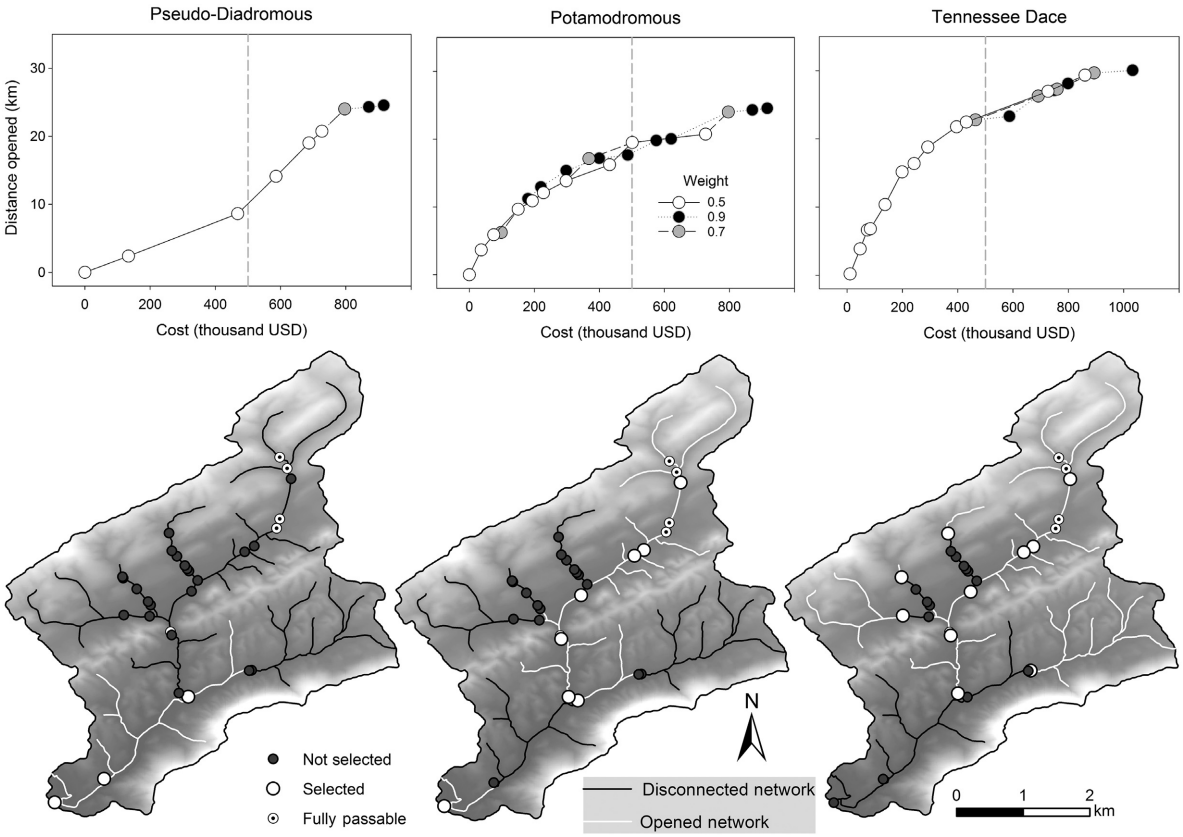

Fig. 2. Three scenarios of barrier removal or replacement in White Oak Creek representing different objectives suited to benefit three fish life histories. Trade-off curves between the total cost and total distance opened are provided for different weighting values (w; see function 2) and different cost thresholds. Barriers selected for removal under a $500,000 total cost constraint (dashed line) are mapped along with stream network opened.

barriers were only selected within the Tennessee dace scenario and were consistently the most upstream barriers on small tributaries. Three barriers were only selected for the diadromous and potamodromous scenarios, and 6 barriers were never selected for RR. The 11 barriers selected within all three scenarios, if removed or replaced, would cost a cumulative total of $796,600 and open 24 km of stream network. This translates to roughly a third of the entire budget for barrier removal to open 73% of the stream network in WOC.

Roanoke River Basin, North Carolina and Virginia

Trade-offs among stream network distance opened, energy lost, and cost of removal were highly variable among the four dam removal scenarios for the RRB (Fig. 4). In the diadromous scenario, the cost of dam removal and losses to hydropower generation rapidly increased with increasing stream distance opened, after which losses to generation were minimal as removals continued. Optimal solutions in the diadromous scenario typically result in the most stream distance opened and largest losses in energy compared to other scenarios, yet the amount of energy lost never exceeded 22% of the total MWh in the basin. By contrast, within the potamodromous and SOC_A scenarios, energy losses remained below 11% while costs continued to increase as more stream distance was opened (Fig. 4). Finally, the SOC_B scenario evaluated distance opened and the number of aquatic species conserved (i.e., objective function) with respect to the cumulative monetary and energy costs of dam removal (Fig. 4). With less than

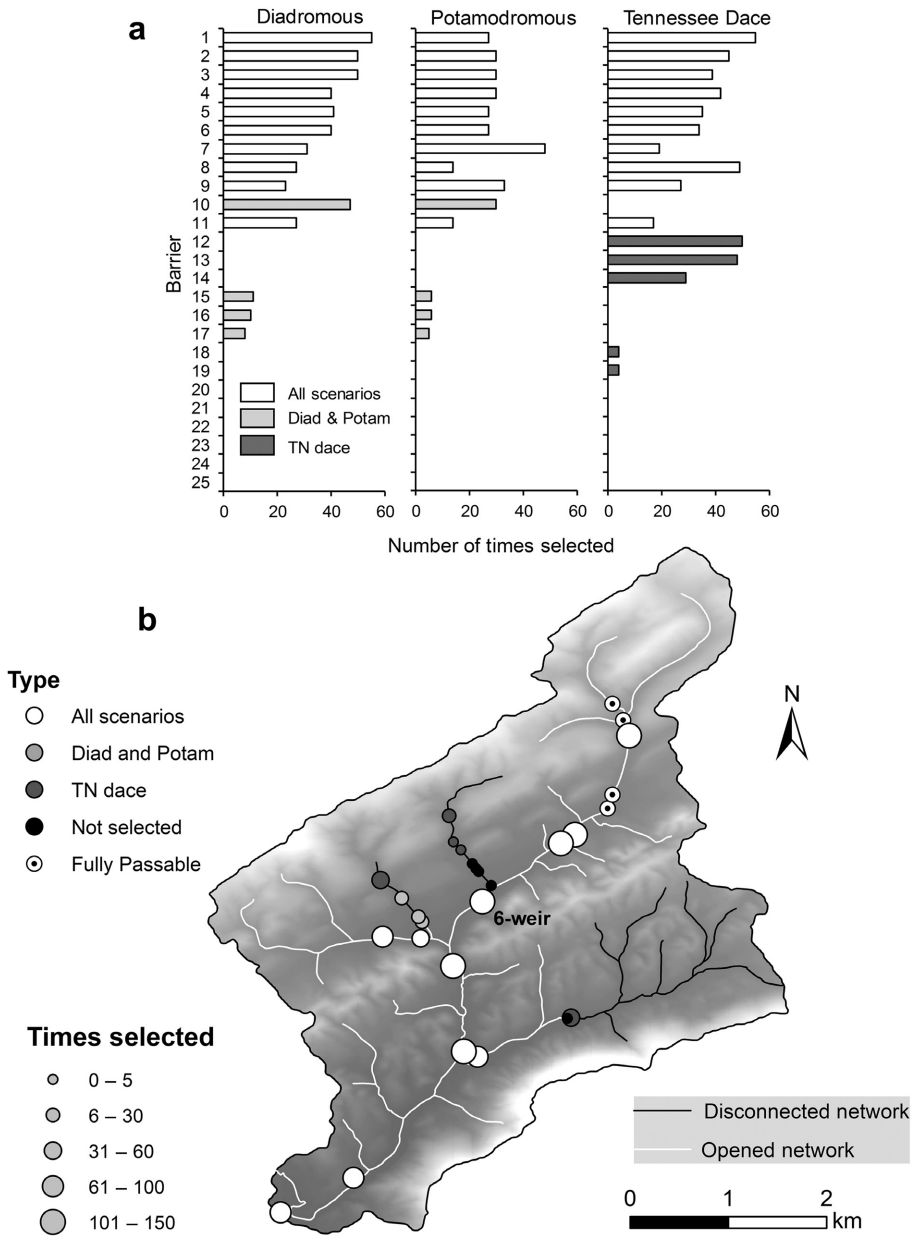

Fig. 3. Potential convergence in barrier removal or replacement scenarios in White Oak Creek. (a) Number of times a given barrier was selected for removal or replacement in each of the three life history scenarios. (b) Barriers are mapped according to the scenarios and frequency in which they were selected for removal or replacement. The barrier "6-weir" was recently selected for removal (see Fig. 10).

$2M expenditures and <1% of energy compromised, dam removals opening ~2500 km of stream habitats would benefit 31 of the 33 species of concern within the basin. Additionally, all 33 species would benefit from a total of $40M expended on dam removals that opened 6700 km of stream and still sacrificed <1% of energy in the basin.

We used a $50M cost threshold to evaluate differences in the degree of stream network opened and energy lost among the four different scenarios in the Roanoke Basin (Fig. 5). In general,

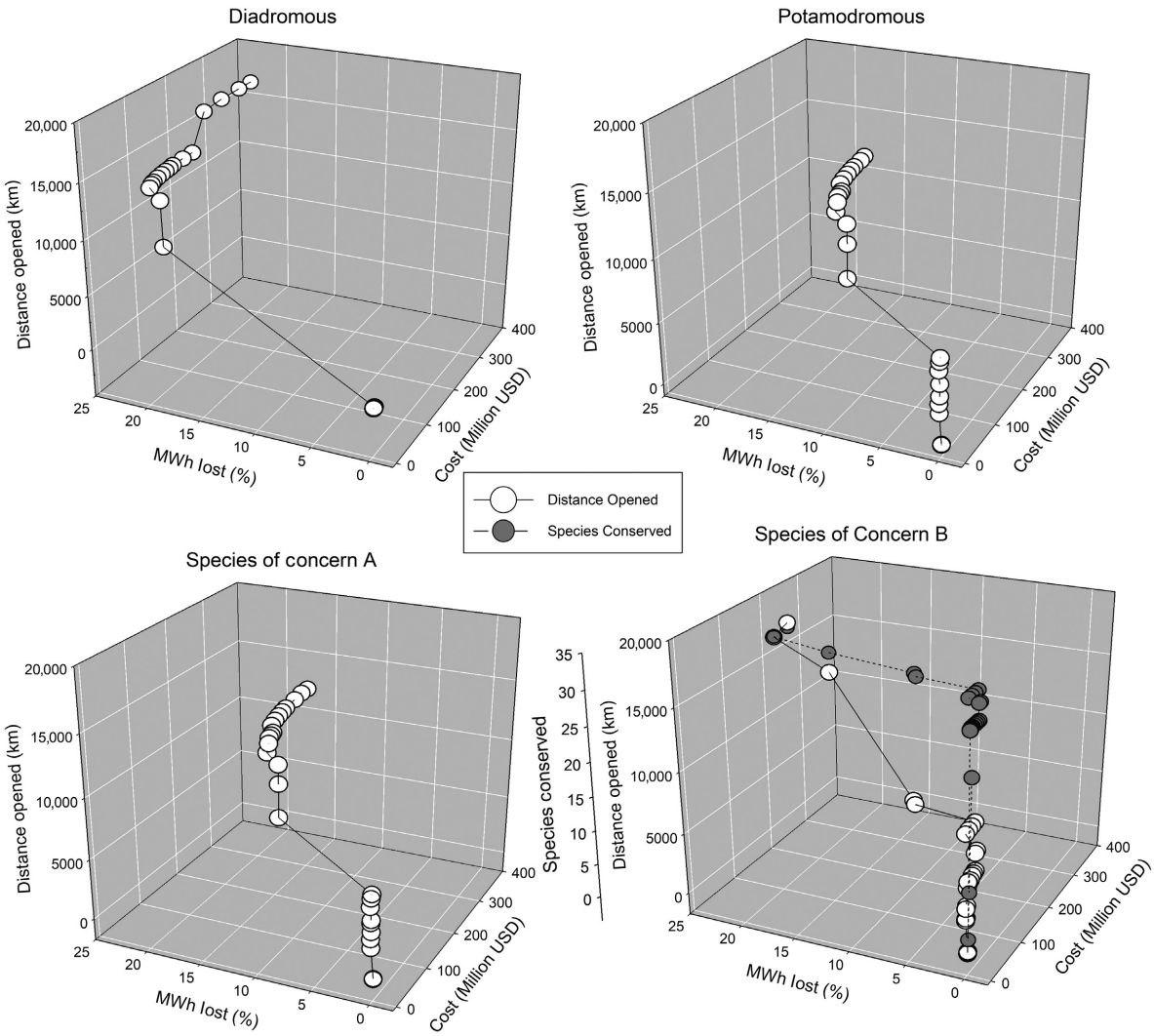

Fig. 4. Trade-off curves between ecological benefits (distance opened or number of species conserved), monetary cost, and energy generation lost following dam removals within four scenarios in the Roanoke River Basin. Four scenarios represent different life history groups.

functional networks upstream of EHDs were far longer than stream distances upstream of NPDs. Expending $50M in dam removals within each of the diadromous, potamodromous, and SOC_A scenarios opened 14,400–16,000 km of stream networks, which represented 71–79% of total stream distance in the Roanoke Basin. However, the energy compromised by those dam removals differed among the three scenarios (Fig. 5). Whereas $50M worth of dam removals in the diadromous scenario compromised 21% of the energy in the basin, removal of dams prioritized under the potamodromous and SOC_A scenarios compromised over 10% of energy. Compared to the other three scenarios, the $50M cost constraint in the SOC_B scenario opened 7730 km of stream (38.3% of total distance) yet compromised <2% of energy generation in the basin.

We applied five Gaussian mixture model algorithms to cluster dams based on their frequency of selection for removal across the four scenarios. The spherical, unequal-volume (VII) model with 7 clusters provided an optimum BIC value for all

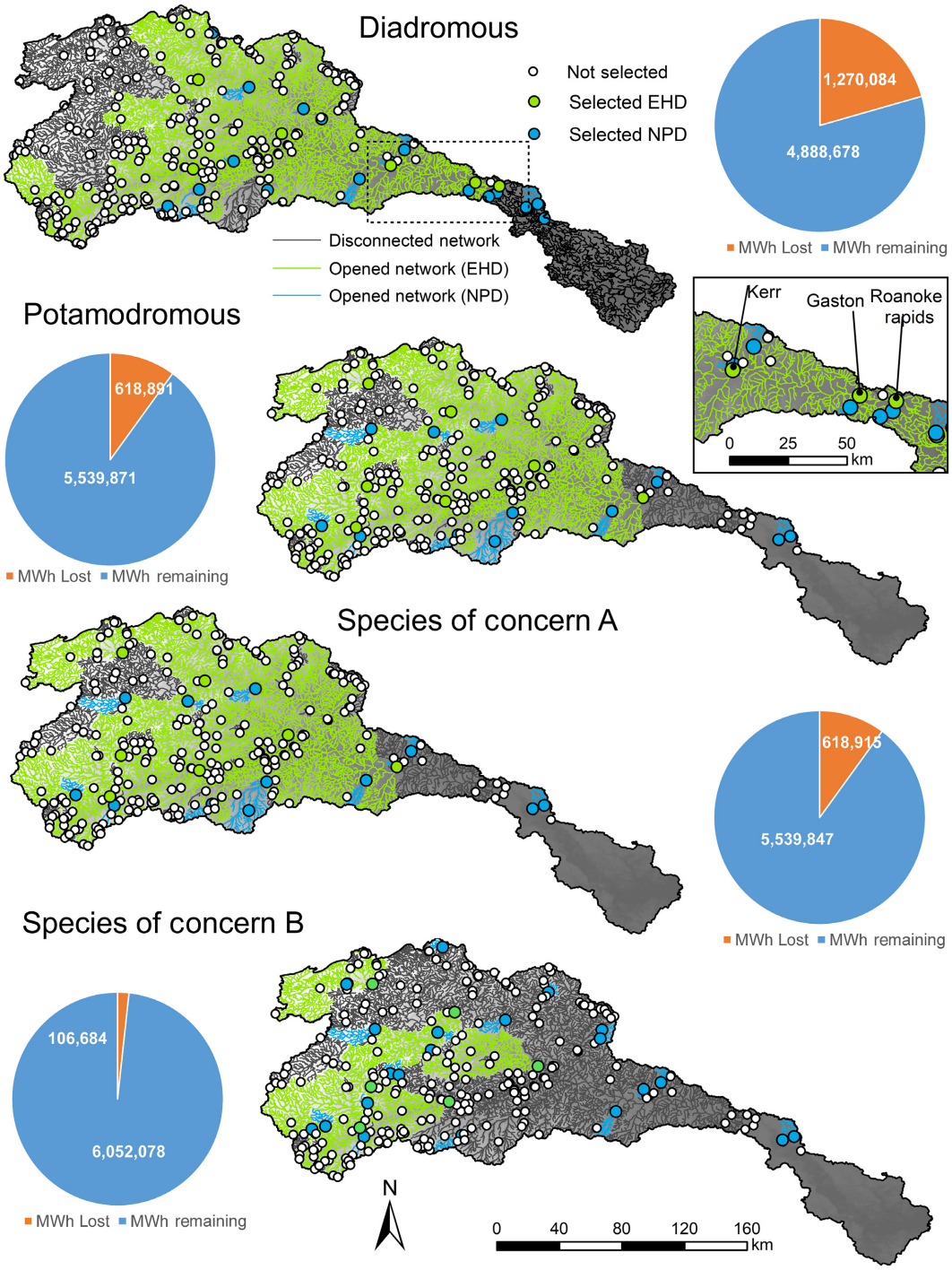

Fig. 5. Maps of existing hydropower dams (EHDs) and non-powered dams (NPDs) selected for removal in the Roanoke River Basin under four life history scenarios with a $50 million cost constraint. Stream networks opened following removal are also mapped. The existing and potential electricity generation lost following dam removal is provided for each scenario.

solutions less than 19 clusters, whereas the maximum BIC value occurred for the ellipsoidal, equal-volume, equal-shape (EEV) model with 20 clusters (Appendix S4). In favor of the more parsimonious solution, we selected the VII model with 7 clusters.

The 7 clusters varied in the total number of dams per group, the frequency in which dams were selected among different scenarios, the cost of removal, the cost of fish passage, energy lost with removal, and distance opened following removal (Fig. 6). Clusters 1 and 2 were the only groups that included membership by both NPDs and EHDs and included dams selected under all four scenarios. All other clusters were solely represented by NPDs prioritized for removal under the diadromous scenario. Whereas clusters 1 and 2 represented the highest energy losses, the cost of removal was comparable with several other clusters; however, the cost of fish passage construction and distance opened was noticeably higher for clusters 1 and 2 (Fig. 6). Generally, the frequency in which a given dam was prioritized for removal was positively related to the functional network upstream of that structure (Fig. 7). Dams within cluster 1, especially EHDs, and dams within cluster 2 represented the longest functional network lengths; however, several members of cluster 2 had relatively short functional networks upstream (e.g., <10 km). Clusters also displayed variable spatial affiliation and aggregation (Fig. 8). Dams not selected under any scenario were distributed in the upper most portion of the RRB. Clusters 4–7 were found in the middle portion of the basin, cluster 3 in the lower portion, and clusters 1 and 2 showed no apparent spatial pattern (Fig. 8). Recursive partitioning models accurately classified 72% of barriers to their assigned cluster using all four variables (Appendix S5: Table S1 and Fig. S1). Clusters with highest misidentification rates were those with low sample sizes (cluster 4 and barriers not selected for removal). When cluster assignments were coarsened into convergent (clusters 1–2), diadromous only (clusters 3–7), and unselected barriers, partitioning models accurately classified 92% of barriers to their correct class using only D_i and the number of species of concern (Appendix S5: Table S2 and Fig. S2). All unselected barriers were incorrectly classified.

For the hybrid scenario, we focused on clusters 1 and 2 because they represented the most frequently selected dams under all scenarios. All NPDs falling into clusters 1 and 2 were removed, whereas fish passage was employed at the 3 EHDs with the largest upstream functional network (Roanoke Rapids, Gaston, and John H. Kerr; Fig. 9). Combined fish passage and removal would cost $177 M USD and open almost 11,000 stream km with no expected losses to existing hydropower and 13.2 GWh of lost potential generation from NPDs (Fig. 9). Generating units at Roanoke Rapids and Gaston are dated 1986–1988 and 1963, respectively, and have never been upgraded, whereas units at Kerr were recently upgraded 2008–2009 (Samu et al. 2018). Generator upgrades at Roanoke Rapids and Gaston could increase generation in the RRB by almost 742 GWh, or a 12% increase in total generation in the basin. However, this would cost an additional $26.8 M USD (Fig. 9).

Discussion

Identifying barriers for removal in streams is increasingly confounded by multiple, potentially conflicting, objectives (Erős et al. 2018). Unfortunately, river managers are faced with the difficultly of reconciling these alternative outcomes, or, potentially more troubling, the reality of choosing one solution over another. Consequently, an increasing number of tools for conducting optimization modeling have come online recently (King et al. 2017, Moody et al. 2017). As noted by King and O'Hanley (2016), the majority of existing modeling frameworks focus only on diadromous fishes (Kuby et al. 2005, King and O'Hanley 2016) and a minority focus on potamodromous (O'Hanley 2011, O'Hanley et al. 2013). Our results showed that different conservation objectives aimed to support varied aquatic species life histories through barrier removal have dramatically different effects on solutions, and our model provides the flexibility to incorporate a variety of life histories. As an example, we observed very different outcomes by changing how ecological benefits were calculated in the objective functions for the SOC_A and SOC_B scenarios, which were otherwise similar in all other respects. Specifically, despite considerable similarities in model construction, treating

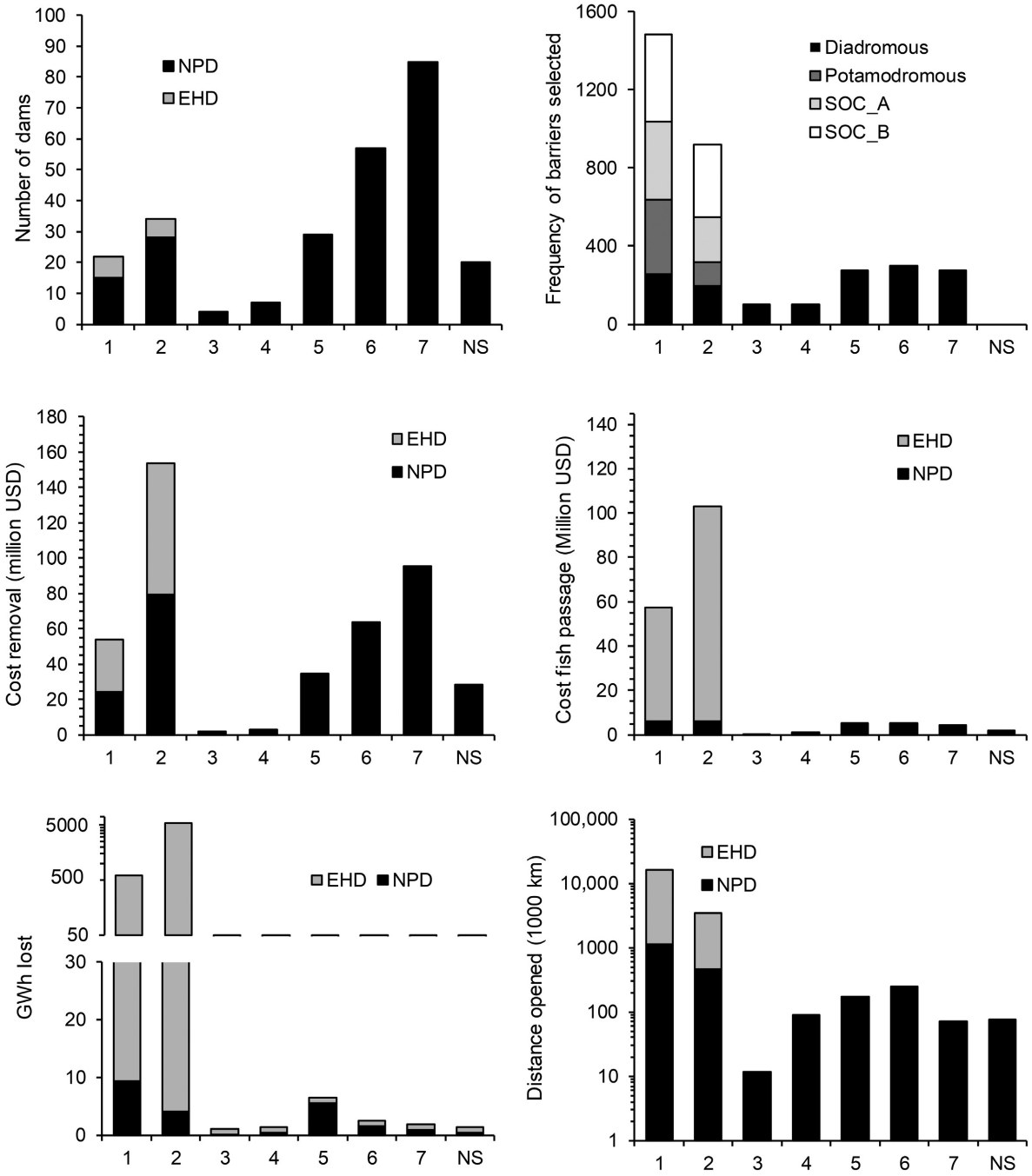

Fig. 6. Characteristics of clusters representing dams selected for removal in the Roanoke River Basin under four life history scenarios and their characteristics. EHD, existing hydropower dam; NPD, non-powered dam; NS, not selected under any scenario.

the number of species of concern that would benefit from barrier removal as the objective instead of the more widely used length of stream reconnected resulted in a much smaller fraction of the river basin being reconnected and large energy savings (Figs. 4 and 5, bottom panels). However, our results also suggest that convergence in competing conservation objectives from barrier

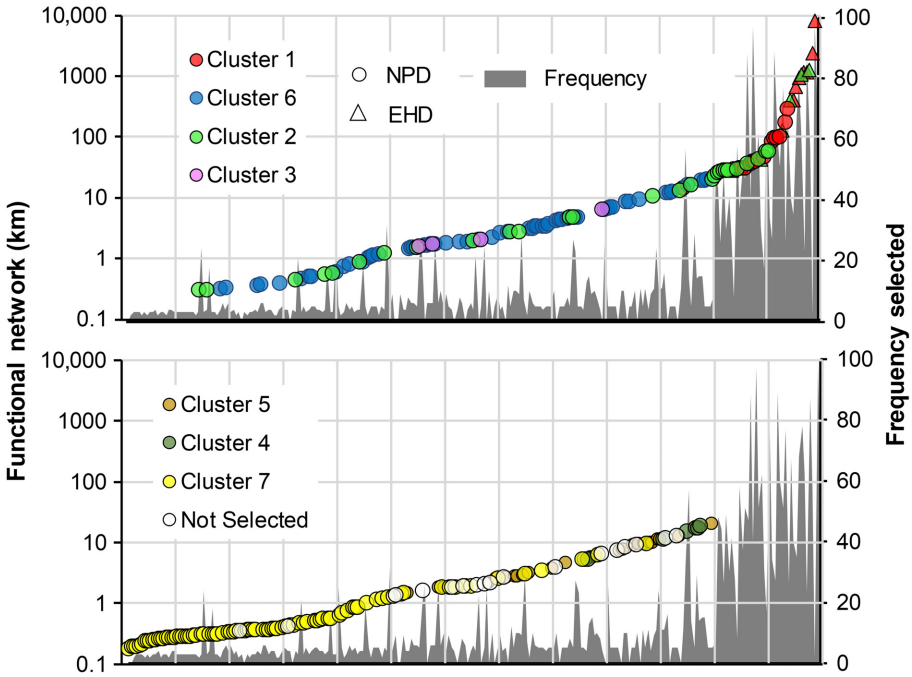

Fig. 7. Barriers ranked according to the length of the upstream habitat they obstruct (i.e., functional network) and plotted according to cluster membership in the Roanoke River Basin. The frequency in which each barrier was selected is in gray shading. EHD, existing hydropower dam; NPD, non-powered dam; NS, not selected under any scenario.

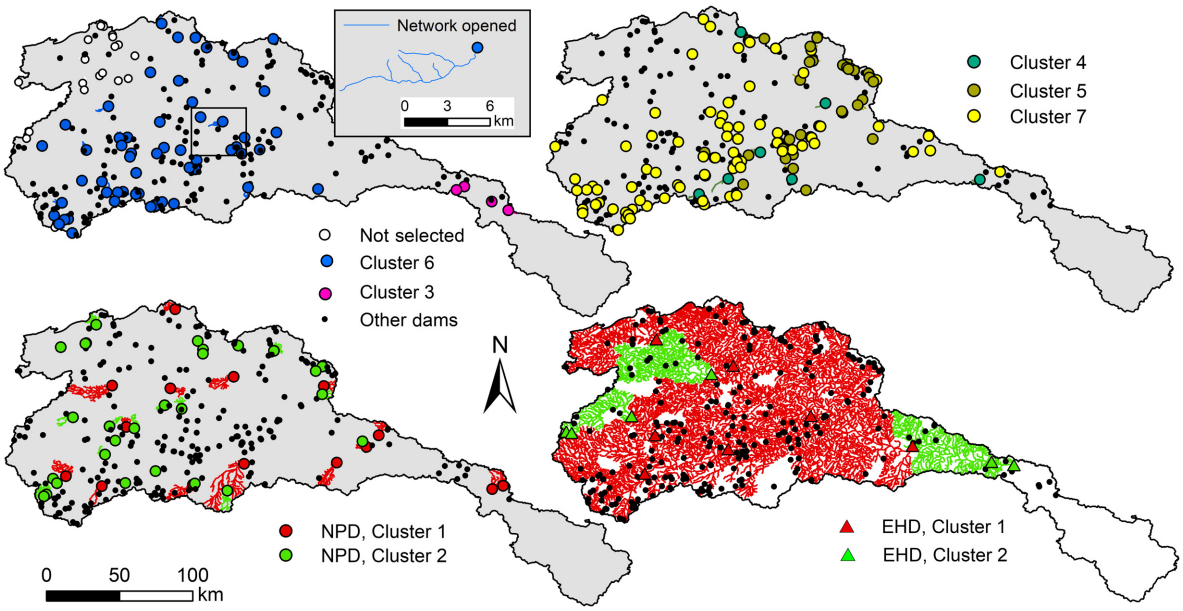

Fig. 8. Barriers and their upstream functional networks are mapped according to cluster membership. Clusters 1 and 2 are separated by those that are NPDs (non-powered dams) and EHDs (existing hydropower dams).

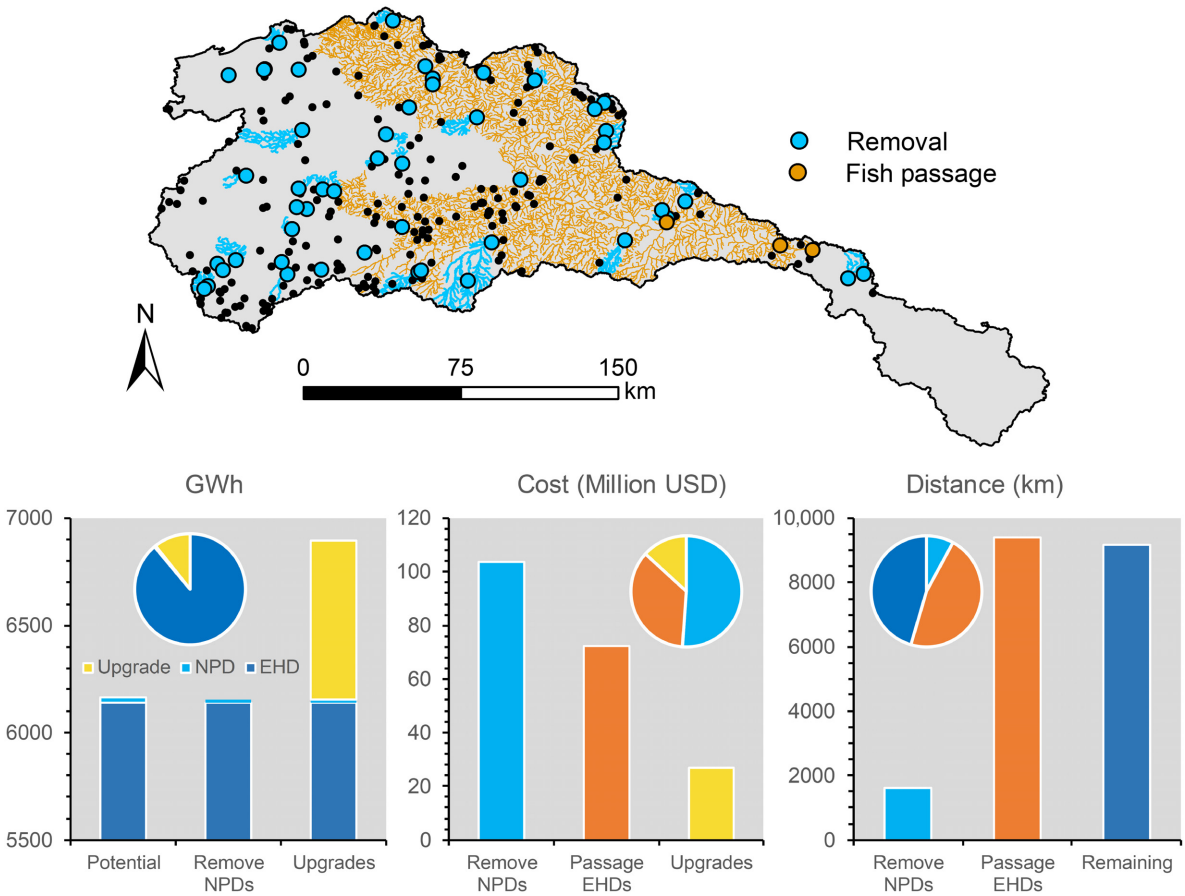

Fig. 9. An example of a hybrid approach combining removal of non-powered dams (NPDs) and provision of fish passage at three existing hydropower dams (EHDs) within clusters 1 and 2 (see Fig. 8). Upgrades in generators at two of the EHDs (Roanoke Rapids and Gaston) are included in the scenario. The costs and benefits of removal, fish passage, and generator upgrades within the scenario are provided and include losses to electricity generation, total monetary cost, and distance opened.

removal can be identified using relatively simple approaches. Furthermore, we show that these general strategies can be identified for all barriers using very simple metrics (i.e., upstream functional network length and presence of species of concern). These results, along with our provision of a familiar platform (Microsoft Excel) for use in optimization modeling, are an attempt to provide conservation decision support tools that are accessible and approachable to managers.

Conflicting objectives can occur within both the cost and benefit components of multi-objective functions, and their specific formulation will influence outcomes. Benefits associated with multiple life history or conservation objectives can be weighed against costs associated with barrier removal, such as monetary expenses, lost revenue from hydroelectric generation, and other purposes of barriers, such as water storage (Kuby et al. 2005). Realistically, selecting barriers for removal is complex and influenced by many ecological as well as socioeconomic factors (Brederveld et al. 2011, Ziv et al. 2012, Erős et al. 2018). Likewise, formulating objective functions and constraints for barrier removal can also be complex, such as conditions for removal imposed by spatial dependencies (e.g., barrier located upstream of another barrier). Such complexities and spatial relationships can be difficult to code within a programming environment, which may

dissuade the optimization of barrier removal for certain life histories or consideration of multifaceted constraints. Our approach highlights that the inclusion of these multiple criteria and constraints is possible with commonly available statistical software and computing power.

Generally, few barrier prioritization approaches explicitly consider ecological attributes, beyond considerations of stream length (Kuby et al. 2005, Kocovsky et al. 2009) or stream patches (O'Hanley 2011; except see King et al. 2017). An exception is Ziv et al. (2012) where the biomass of migratory fish lost to scenarios of future hydropower production in the Mekong River Basin was dependent upon a fish migration model, which relied on locations of migratory fish, migratory effectiveness (i.e., distance to floodplain habitats), connectivity to floodplain habitats, and hydroperiod seasonality. Despite the incorporation of ecological complexity, Ziv et al.'s analysis only considered a single life history strategy, that is, long-distance potamodromous migratory species traversing from upstream dry-season habitats to flooded downstream habitats in the wet season. Hence, consideration of other life histories may have influenced optimization results (see SI discussion; Ziv et al. 2012). In the WOC case study, we show that including alternative life history strategies (e.g., headwater species) beyond considerations of diadromous and long-distance potamodromous life histories can influence outcomes. Specifically, solutions benefiting Tennessee dace prioritized barriers that disconnected small tributaries from mainstem habitats, whereas the generic potamodromous scenario sought to maximize patch area in the core of the watershed. In the RRB case study, we included the occurrence of imperiled species in optimizations, first adding species occurrence as a simple constraint to the generic potamodromous scenario (SOC_A) and second by changing the objective function to consider the number of species benefitting from barrier removal (SOC_B). The SOC_A scenario involving only occurrence of imperiled species produced results very consistent with the potamodromous scenario, perhaps because the probability of imperiled species occurrence increases with habitat fragment size. However, including the number of imperiled species conserved with barrier removal (SOC_B) resulted in solutions unique from those developed with more traditional potamodromous and diadromous scenarios because known regional hotspots in imperiled species diversity were highlighted rather than streams that might be inhabited (Fig. 5).

Convergence in divergent barrier removal scenarios

Because the majority of studies evaluating U.S. dam removals have been opportunistic (O'Connor et al. 2015*a*, Bellmore et al. 2017), we presume that the selection of barriers for removal has also been conducted in a similar fashion (McKay et al. 2017). Two barrier removal projects, one in each of our watersheds, illustrate opportunistic approaches to stream connectivity restoration. In July 2016, one of the weirs identified in our study (barrier 2, also called 6-weir; Fig. 3) was removed, primarily to improve fish passage in WOC, but also because the structure had become dilapidated and inoperable due to a head-cut and leakage at its base (Fig. 10). Despite the fact that 6-weir was the 2nd most selected barrier for removal out of all our scenarios for WOC (Fig. 3), no decision support tools were consulted prior to facilities management taking actions to remove the weir. As another example, the 7.6-m Power Dam on the Pigg River, Virginia, was selected under all scenarios (cluster 1) in our assessment, and in 2016, it was removed (Fig. 10) opening 171 km of upstream habitat. In 2005, the U.S. Fish and Wildlife Service and multiple partners developed a plan for removing Power Dam, primarily due to the presence of the federally endangered Roanoke logperch (*Percina rex*; Byrd 2018). While preparations for Power Dam removal required considerable planning, permitting, and funding acquisition, there were no documented basin-scale prioritizations or cost-benefit analyses that informed the decision process.

Of course, the above strategies are seemingly at odds with the barrier prioritization literature, which emphasizes the need to holistically evaluate costs and benefits of removals within entire watersheds or basins (O'Hanley 2011, Jager et al. 2015, Neeson et al. 2015). Some have suggested shifting from localized opportunistic approaches to nationwide adaptive management strategies

Fig. 10. Examples of barrier removals in White Oak Creek (a, b) and the Roanoke River Basin (c, d). In White Oak Creek, 6-weir (a) before and (b) after removal. Power Dam on the Pigg River (a) during and (b) after removal. Photo credits: (a, b) Trent Jett (Oak Ridge National Laboratory); and (c, d) Bill Tanger (Friends of the Rivers of Virginia).

to prioritize dam removals and associated monitoring studies (Bellmore et al. 2017). The scale of this approach would undoubtedly require understanding and reconciling divergent ecological and socioeconomic objectives across geographies (Neeson et al. 2015).

Although trade-offs and compromises are obvious solutions to competing objectives, our approach emphasizes the reality of commonalities among even divergent objectives. For example, in WOC, some barriers would benefit fish habitat connectivity regardless of life histories (i.e., barriers were selected under multiple objectives). These are obvious targets for opportunistic RR associated with general habitat connectivity improvement. That is, targeting specific barriers is generally difficult because extenuating circumstances may prevent removal (e.g., insufficient funds mobilized contaminants; Stanley and Doyle 2003). Alternatively, a pool of potential barriers that satisfy multiple conservation objectives can provide a 1st-order prioritization of barrier removals as a foundation for strategic approaches for resource expenditures.

Our work emphasizes emergent properties in barrier characteristics that can be used to guide basin-scale barrier removal. In the RRB, we identified eight clusters of barriers that represent different types of strategies for approaching basin-scale removal. Although clusters were purely based on the frequency of their selection for removal, clusters varied in physical characteristics, such as height and generation, and physiographic properties, such as river size and length of upstream functional network. The first most notable group was 20 barriers that were never selected for removal among any of the scenarios. These relatively small structures (average height, 10 m) occurred on small streams and were geographically clustered in the northwestern

headwaters of the RRB (Fig. 8, upper left). The total stream distance upstream of these structures represents <1% of the total stream length in the basin, yet the total cost of removing these structures would exceed $28M USD. Our analysis suggests that these barriers lead to little benefits in terms of increasing fish habitat access and resources should be preserved for removal of barriers in other clusters.

The remaining clusters can be grouped into four strategies for assisting in prioritizing barrier removal. Commoners (clusters 3, 6, and 7), comprised entirely of NPDs, represented 56% of barriers in the basin and had short individual upstream networks (average, <3 km each). Collectively, their removal would open intermediate amounts of upstream habitat to diadromous fish but would require significant resources ($160M USD). These barriers represent ubiquitous (common) features on the landscape and may be commonly targeted for opportunistic removals to benefit diadromous fish because of their abundance. Rarities (cluster 4) consisted of 7 small NPDs (mean height, 10 m) that would cost relatively little to remove ($3.2M) but would yield disproportionally large upstream habitat in return (88 km). On an individual basis, these rarities represent rare but highly beneficial opportunities to reconnect relatively large amounts of habitat for little monetary cost (27 km per million USD) or little cost to energy losses. Because of limited occurrences, however, they do not present large cumulative distances of open habitat once removed compared to the commoners. Prospects (cluster 5) consisted of 29 NPDs on medium-sized streams with the potential to generate 5.5 GWh of electricity annually or enough power for 400 homes (EIA 2018). These prospects represent potential gains in electricity generation if left in place, or relatively expensive removal opportunities that would return little upstream habitat (5 km per million USD). Additionally, we estimate that the capital costs of retrofitting these structures for hydroelectric production and constructing a fish passage facility would be <50% of dam removal costs. Finally, goliaths consisted of large (cluster 1) or very large (cluster 2) EHDs and NPDs that, if removed, would open 97% of stream distances in the basin and benefit all life histories, but only after very high monetary costs ($3–4M USD each or $208M USD total) and significant losses to potential or realized energy production. These goliaths are commonly identified as contributing to ecological degradation and were selected across multiple life histories in our study, but these same structures provide many services. Of these, 43 NPDs, if removed, would only open 8% of stream distance in the basin or, alternatively, could provide 950 homes with electricity.

While the capital costs of powering these NPD facilities and adding fish passage facilities would be <36% of the cost of removal, the ecological benefits of fish passage facilities are unlikely to meet those of dam removal (Silva et al. 2017). Whereas the complete or partial removal of obstacles reinstates natural processes associated with the longitudinal and lateral connectivity of river systems, fish passage facilities may be ineffective at passage and typically consider only the upstream portion of migration, not the downstream portion, and facility designs may only benefit some species, not the entire community (Silva et al. 2017). In contrast to NPDs, the capital cost of fish passage at the largest EHD facilities is estimated to be higher than removal costs, based on the cost curves we used. It should also be noted that all NPDs in the RRB, if powered, would only constitute <1% of total hydropower generation provided currently by EHDs in the RRB. Efficiency upgrades to EHD facilities likely comprise a much greater opportunity for increasing hydropower generation (Johnson et al. 2013).

Ultimately, consideration of barrier removal will consider the source and magnitude of hydropower potential in relation to improvements in habitat through removal or fish passage. Decisions regarding what to do with commoners, rarities, prospects, and goliaths will ultimately be dependent upon objectives identified by stakeholders interested in restoring rivers within a basin. Our approach highlights one of many frameworks for quantitatively comparing contrasting objectives and identifying potential convergences. We do not advise using the framework to focus all removal efforts on one strategy over another, but rather using the strategies to communicate the costs and benefits of barrier removals including finding convergence where barrier removals would address multiple alternative objectives. Furthermore, optimal removal strategies for a given basin may occur across

multiple strategies (e.g., stratified design) and may incur a combination of approaches for opening stream habitats, such as removal and passage (Opperman et al. 2011). As an example of one such approach, we provided a hybrid approach (Fig. 9) focused on the goliath strategy, as removing or retrofitting these barriers could benefit multiple life histories. We found that removing goliath NPDs and providing fish passage at the three down-most EHDs would open almost 55% of stream length in the RRB with very little cost to existing energy and little impact on potential energy sources. Furthermore, if upgrades to generators at Roanoke Rapids and Gaston were conducted, overall generation in the basin could increase by 12%. However, undertaking such an option would require significant monetary costs, totaling $177M USD alone for removal and fish passage and an additional $26.8M USD to upgrade generators at the two facilities. The costs of removal and passage alone represent 37% of the total cost of removing or retrofitting all barriers identified in the study. Hence, our clusters provide a flexible template to bridge purely opportunistic style approaches to dam removal with decision support determined via optimization modeling.

Comparing convergent solutions with sustainable basin design principles and current restoration efforts

Holistic considerations of societal and ecological demands for river systems are more likely to be addressed in decisions at the basin level (Opperman et al. 2011, Neeson et al. 2015), as opposed to decisions at local or geopolitically constrained levels. We further suggest that holistic decision-making regarding barrier removal and siting would consider a diverse set of ecological objectives, such as meeting multiple life history needs. In a review of sustainable river basin designs, Jager et al. (2015) provided several design principles for the siting or removal of dams aimed to increase the overall ecological sustainability of rivers: (1) Preserve free-flowing habitats in core mainstem areas by concentrating barriers within a subset of tributary watersheds, (2) distribute reserves among remaining tributary watersheds to preserve connectivity between the mainstem and selected smaller watersheds, (3) ensure habitat between dams is sufficient to support reproduction and recruitment, and (4) conduct spatial decision assessments at the large-basin scale. These principles provide a template for comparison to our scenarios and their convergence, or lack thereof.

Although WOC is not a large basin, Jager et al.'s other three principles still apply. Whereas each WOC scenario independently did not meet the principles suggested by Jager et al. (2015), the convergent solution, that is, barriers selected at least once under all scenarios, aligns well with sustainable basin design criteria. For instance, the diadromous scenario focused on opening barriers in the downstream sections of WOC, thereby avoiding uppermost tributary connections. In contrast, the potamodromous scenario prioritized barriers that, if removed, would provide patch connectivity in the core of WOC, not necessarily downstream habitats or small tributaries. Barriers identified under the convergent solution, if removed or replaced, would open the entire mainstem, reconnect two tributaries, and concentrate barriers in the other remaining tributaries (Fig. 3). Among the barriers selected under all scenarios, barrier 2 (also called 6-weir) was the 2nd most selected barrier for removal out of all scenarios (Fig. 3). In July 2016, the weir was removed to improve fish passage in WOC, as the structure has become dilapidated and inoperable due to a head-cut and leakage at its base (Fig. 10).

Individual scenarios in the RRB would only meet some, not all, of the criteria. However, the convergent scenario for the RRB (i.e., goliath dams, clusters 1 and 2) would meet the all the criteria of the sustainable river basin design. Particularly, 10 of the goliath EHDs in cluster 1, if removed, would open considerable stream distance in the basin, but would come only at great cost. Alternatively, our hybrid approach could provide a more feasible alternative to increase connectivity and meet sustainable basin designs, yet not compromise large hydropower generation in the basin. Indeed, fish passage efforts are underway already within the RRB. Under a 2005 license order approval and settlement agreement, Dominion Power was required to develop a fishway for American eel and trap/transport facilities for American shad (*Alosa sapidissima*) at Roanoke Rapids and Gaston facilities (FERC 2005).

Limitations and Conclusions

When conducting barrier removal assessments, there are generally several common limitations, and our study is no exception. First, most inventories of barriers within a given region are incomplete due to coarse mapping efforts that miss smaller impoundments (Ignatius and Jones 2010). While this is likely true for the RRB, the barrier assessment in WOC was extensive and included manually traversing the watershed to ensure a complete inventory (McManamay et al. 2016). Another consideration is that we scaled hydrography of our analyses to match the respective objectives for barrier removals in WOC and the RRB. For instance, the WOC assessment relied on high-resolution hydrography to assess small barrier removal (culverts and weirs) whereas coarser hydrography was used in the RRB to prioritize larger dam removals, primarily structures ranging from 3 to 82 m in height. It should be noted, however, that even high-resolution hydrography in WOC will exclude ephemeral systems and potentially underestimate stream habitat lengths.

Another limitation is that our cost estimates may be conservative and may not accurately reflect all associated costs beyond the initial capital required for removal, fish passage, or replacement. Long-term costs include those associated with operations and maintenance of power plants and fish passage facilities. Based on documented costs of removal of two dams in the RRB, our cost model (Eq. 1) is within 10–30% of the actual removal costs. Brantley Dam removal on the Dan River was removed for <$30,000 (Bloom 2011) whereas our model estimated $27,400 for a structure of 1.8 m. Power Dam removal cost $500,000, although total project costs were $1 M, including monitoring studies (Fabris 2017). Our model estimated removal costs for Power Dam at $360,000. Beyond monetary costs and losses to energy, there are many other potential costs of lost services provided by barriers not considered in our analysis, such as loss in water storage, recreation, and flood control. Additionally, there are numerous other ecological considerations besides the ones considered herein that could influence the selection of barriers for removal, such as spread of invasive species (Milt et al. 2018); mechanistic assessments of ecological benefits, that is, recruitment (Ziv et al. 2012); or habitat quality (Kocovsky et al. 2009).

In both of our case studies, we developed general strategies aimed to support one or more divergent conservation objectives through barrier removal or replacement solutions. We identify solutions that would meet ecological sustainability design criteria, in light of realistic cost constraints. In some cases, removing all barriers to meet sustainability criteria would likely be infeasible, or in the least, come at very high costs, both monetarily and as losses to societal services provided by structures. For instance, in WOC, the risk of mobilizing decades worth of contaminated sediments at the downstream-most dams makes removing these structures unlikely. Likewise, removal of the three downstream-most dams in the RRB would constitute a significant regional loss in renewable energy, particularly a scenario where renewables are incapable of meeting peak energy demands. Without considerable energy storage capabilities, wind and solar resources lack the flexibility and reliability to meet these energy demands and offset the role of hydropower in balancing the electricity grid (Budischak et al. 2013, Matek and Gawell 2015, Solomon et al. 2017). Despite these limitations, however, we believe sustainable basin design criteria can be met through hybrid removal, passage, and powering options (e.g., Opperman et al. 2011, Jager et al. 2015). Ultimately, our framework can be used to identify convergent solutions to strategically approach barrier removal, including holistically communicating the associated benefits and costs.

Acknowledgments

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2596/full