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Prioritizing Stream Barrier Removal to Maximize Connected Aquatic Habitat and Minimize Water Scarcity

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Research Impact Statement: Prioritizing stream barrier removal using dual-objective optimization quantifies tradeoffs between quality-weighted, connected fish habitat and water scarcity costs of reduced water deliveries to cities.

ABSTRACT: Instream barriers, such as dams, culverts and diversions, alter hydrologic processes and aquatic habitat. Removing uneconomical and aging instream barriers is increasingly used for river restoration. Historically, selection of barrier removal projects used score-and-rank techniques, ignoring cumulative change and the spatial structure of stream networks. Likewise, most water supply models prioritize either human water uses or aquatic habitat, failing to incorporate both human and environmental water use benefits. Here, a dual-objective optimization model identifies barriers to remove that maximize connected aquatic habitat and minimize water scarcity. Aquatic habitat is measured using monthly average streamflow, temperature, channel gradient, and geomorphic condition as indicators of aquatic habitat suitability. Water scarcity costs are minimized using urban economic penalty functions while a budget constraint specifies the money available to remove barriers. We demonstrate the approach using a case study in Utah's Weber Basin to prioritize the removal of instream barriers for Bonneville cutthroat trout, while maintaining human water uses. Removing 54 instream barriers reconnects about 160 km of quality-weighted habitat and costs approximately US$10 M. After this point, the cost effectiveness of removing barriers to connect river habitat decreases. The modeling approach expands barrier removal optimization methods by explicitly including both economic and environmental water uses.

Keywords: connectivity, optimization, restoration, river, river network, trout

INTRODUCTION

Dams, culverts and diversions, collectively referred to as instream barriers, are economically-important for water supply and conveyance, but negatively affect river ecosystems and disrupt hydrologic processes. Instream barriers change chemical, physical and biological properties of rivers by altering stream temperature, dissolved oxygen, discharge, river depth, sediment transport and movement of native and non-native species (O'Hanley, 2011). Removing uneconomical and aging instream barriers to improve aquatic habitat connectivity is a technique increasingly used to restore river habitat (Stanley and Doyle, 2003; Magilligan *et al.*, 2016). Including both human water demands and aquatic habitat objectives in research and modeling advances understanding of environmental-economic tradeoffs to restore suitable habitat connectivity while managing competing human water uses (Null *et al.*, 2014).

Small barriers like diversion dams, weirs, and culverts fragment habitat patches and inhibit species' migration and movement. They reduce genetic variability between populations (Pringle, 1997; Compton *et al.*, 2008; Peterson *et al.*, 2013). Many past barrier removal studies focused on identifying individual barriers to remove using a score-and-rank technique, which scores physical, economic or ecological attributes of barriers, then ranks them for potential removal. Scoring-and-ranking is straightforward and simple, but does not consider the cumulative hydrologic, habitat, or ecological effects of removing multiple barriers within the stream system (O'Hanley and Tomberlin, 2005; Kemp and O'Hanley, 2010; O'Hanley, 2011). Barrier removal systems modeling has focused on maximizing aquatic habitat connectivity but ignored economic benefits of dams, like water supply reliability, hydropower generation, recreation, or flood damage reduction. When costs were included, they were for dam removal, remediation (Zheng and Hobbs, 2013; Reagan, 2015; King and O'Hanley, 2016), or occasionally habitat restoration (Null and Lund 2012). Conversely, water resources systems models commonly include economic objectives, but represent environmental criteria as constraints, removing them from decision-making (Cai et al., 2003; Jager and Smith, 2008; Null 2016).

Several studies have represented instream habitat and economic water supply objectives, although habitat was typically modeled simplistically as accessible drainage area, river length, or passability of barriers at different flows (Kuby et al., 2005; Null et al., 2014; Neeson et al., 2015). Kuby et al. (2005) quantified and visualized trade-offs between salmonid migration, hydropower generation, and water storage. Stream length was summed to quantify habitat, assuming that all connected river segments provided suitable habitat. Zheng et al. (2009) and Zheng and Hobbs (2013) included economic losses from barrier removal and invasive species control, but water reliability was not included as an objective. Null et al. (2014) minimized water scarcity from large dam removals in California. Tradeoffs were evaluated between economic scarcity costs of dam removal and environmental benefits of suitable upstream habitat; however, aquatic habitat was not included directly in the optimization model. Most recently, Neeson et al. (2015) used a return-on-investment optimization approach to analyze gains of barrier removal at different spatial and temporal scales. Their project is noteworthy because cost efficiency of barrier removal was evaluated basin wide and through time to understand the significance of allocating funding for restoration projects. Their study did not include economic losses from lost water deliveries.

To consider both water scarcity costs and aquatic habitat gains when prioritizing barrier removal, we developed a dual-objective optimization model to evaluate barrier removal benefits given economic and environmental objectives, and account for the interconnected, spatial structure of a river network. Dual-objective optimization mathematically maximizes or minimizes specific objectives, resulting in a Pareto-frontier tradeoff curve, where points on the curve are efficient solutions (Pareto, 1964). Here, the environmental objective is maximized to benefit aquatic habitat connectivity for trout, using monthly average streamflow, water temperature, channel gradient, and geomorphic condition as indicators of aquatic habitat suitability. Habitat suitability is multiplied with reach length to determine reach quality-weighted habitat. An adapted version of the Integral Index of Connectivity (IIC) (Saura and Pascual-Hortal, 2007) calculates improved connectivity between quality-weighted habitat from removing barriers. The economic objective is minimized to limit water scarcity costs using urban economic penalty functions. We use the weighting method to combine two objective functions into a single objective optimization problem. Weights on each objective vary between model iterations to produce the Pareto-frontier curve. A budget constrains barrier removal costs and limits the number of barriers to remove.

Our approach is novel because it simultaneously considers human water uses and quality-weighted fish habitat connectivity for a large number of barriers and potential barrier removals at the watershed-scale. It provides information for managing competing human and environmental water demands, in this case, by prioritizing instream barrier removal to improve accessibility to fish habitat at the least cost for people. The model is applied to northern Utah's Weber Watershed. We focus on restoring habitat for protected

Bonneville cutthroat trout (*Oncorhynchus clarki Utah)* as an indicator of high quality, connected aquatic habitat in the Weber Basin, although the model formulation is generalizable to other basins. This paper begins with a description of the Weber Basin, followed by modeling methods and assumptions including aquatic habitat suitability classification, quality-weighted habitat connectivity, barrier passage ratings, cost of barrier removal, and water scarcity cost estimates from economic penalty functions. Results and discussion focus on tradeoffs between barrier removal costs, water scarcity costs, and quality-weighted habitat connectivity. The paper ends with a discussion of model limitations, followed by a summary of the five main conclusions of the paper.

STUDY SYSTEM AND BACKGROUND

Utah's Weber River flows approximately 200 km from the high Uintah Mountains to Great Salt Lake (Figure 1). The watershed is about 6,400 square kilometers (km^2). Snowmelt from the Wasatch and Uintah Mountains is the primary source of water. The basin has a montane to semi-arid environment and receives about 380 - 430 millimeters (mm) of precipitation per year (SWCA, 2014). The Weber Basin model includes 348 barriers, defined as any unnatural instream structures, 66 in the mainstem Weber River and 282 in tributaries.

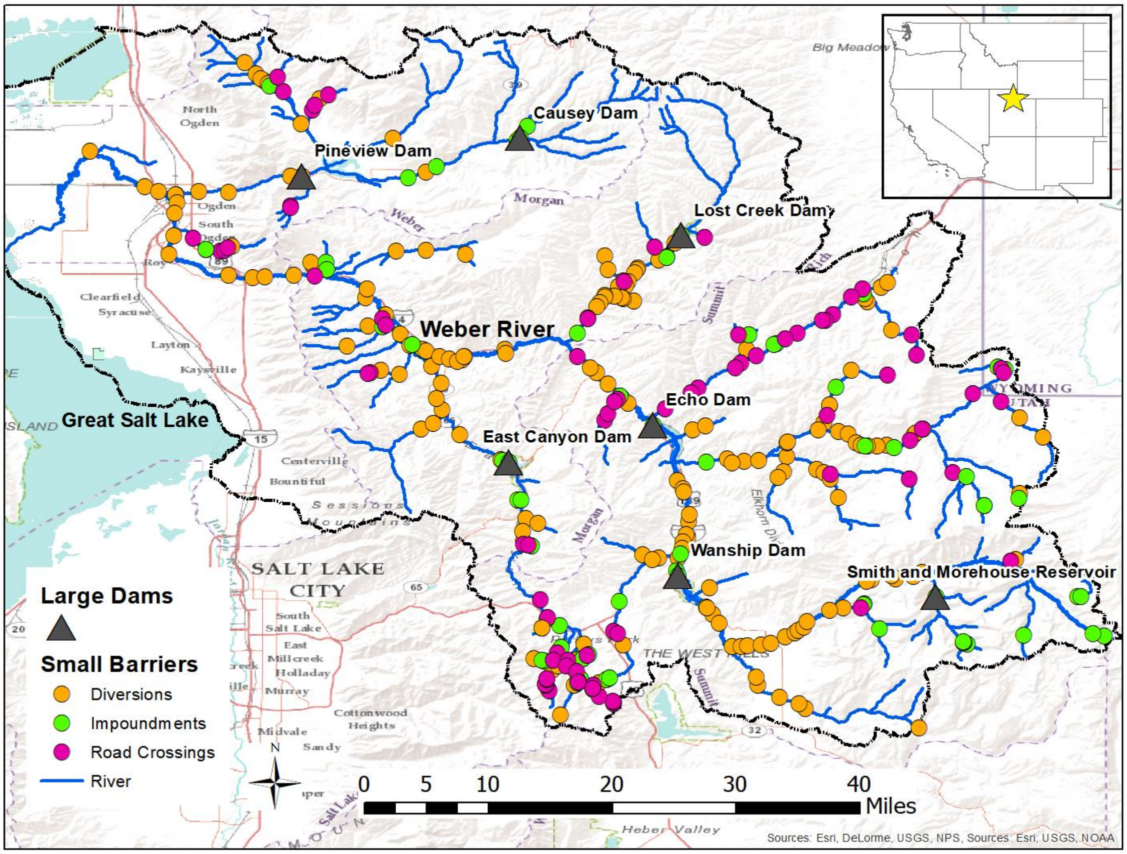

FIGURE 1. Weber Basin (Data sources: U.S. Geological Survey, 2013, accessed 12/2015; Esri, 2017, accessed 01/2018). Dots represent small barriers such as diversion dams, impoundments, and road crossings. Large dams are represented by triangles. Stream barriers were combined to develop a barrier database (Data sources: NHD, accessed 12/2015, U.S. Geological Survey, 2013, accessed 12/2015; Trout Unlimited, 2014, accessed 03/2016; National Inventory of Dams, accessed 03/2016).

The Weber River is highly regulated. Discharge averages about 12.5 cubic meters per second (m^3/s) near the outlet to Great Salt Lake but would be considerably higher without consumptive water uses (Weber River Near Gateway USGS Gage 10136500) (Wurtsbaugh *et al.*, 2017). Currently, the Weber River supplies about 98.2 million cubic meters (Mm^3) of water to municipal and industrial water users each year and 266.4 Mm^3 annually for irrigation (Weber Basin Water Conservancy District, 2010). The basin provides water for over 500,000 people along the Wasatch Front and this population is projected to nearly double by 2050 to one million people (Utah Foundation, 2014).

Bonneville Cutthroat Trout Habitat

The Weber River historically supported healthy populations of Bonneville cutthroat trout. Altered environmental conditions reduced access to suitable habitat and competition with nonnative species have led Bonneville cutthroat trout to be listed as a "conservation species" in Utah (Budy et al. 2007). This means that Bonneville cutthroat trout are protected under a multi-state conservation agreement to eliminate threats to ensure long-term survival of populations and avoid listing under the Endangered Species Act (Webber et al., 2012). Considering the conservation goal of this species, restoring connectivity of suitable habitats is essential to sustain and enhance viable Bonneville cutthroat trout populations.

Cutthroat trout prefer clear, cold water and complex habitats with sufficient depth for migration, depending on life stage (Budy *et al.*, 2007). Annual spawning for Bonneville cutthroat trout occurs in spring and into summer at higher elevations (Bennett *et al.*, 2014). Trout prefer water temperatures under 15 °C (Bear *et al.*, 2007) but can survive in temperatures over 22 °C and potentially up to 26 °C for short periods of time (Schrank, *et al.*, 2003). Ideal water depth for adult cutthroat trout ranges between 0.4 and 0.7 m, and 0.3 to 0.6 m for juveniles in low velocity streams (Kershner, 1992; Braithwaite, 2011).

Movement of Bonneville cutthroat trout are greatest in spring, moving distances up to 82 km per season, although the majority of fish relocate less than 10 km within the river. Summer and winter movement is generally within 1 km but, at times, cutthroat trout move up to 22 km (Schrank and Rahel, 2004; Colyer *et al.*, 2005; Carlson and Rahel, 2010). Habitat fragmentation between metapopulations in the Weber Basin limits population dispersal and prevents access to preferred spawning reaches and other suitable habitat (Budy *et al.*, 2014). Connectivity between habitats is important for access to suitable habitat, but also to maintain genetic variability and exchange between populations (Budy *et al.*, 2007; Budy *et al.*, 2014). Disconnected subpopulations become isolated, increasing potential extinction risk (Hilderbrand and Kershner, 2004).

Barrier Removal Decision-making in the Weber Basin

Weber Basin stakeholders have implemented fish passage projects for river restoration. In 2012, the National Fish Habitat Association listed the Weber River as "Water to Watch" because of recent efforts to reconnect about 27 km of habitat by building a fish passage structure on a mainstem river barrier and reconstructing two previously impassable culverts (National Fish Habitat Partnership 2012). Trout Unlimited has an ongoing project assessing potential fish passage barriers using aerial photography and water rights data (Paul Burnett, Per. Comm., 2015). Given the scope and magnitude of barrier effects on river habitat and aquatic ecosystem health, removing barriers offers an opportunity to restore aquatic habitat connectivity (Stanley and Doyle, 2003; Magilligan *et al.*, 2016). However, the number of barriers and restoration options, large network, and competing water management objectives make it challenging to identify which barriers to remove, ultimately hindering decision-making.

METHODS

We developed a dual-objective optimization model to prioritize barrier removal (Figure 2). In this section, we first describe the mathematical model formulation for each objective, as well as how we combine the two objectives into a single objective optimization problem using weights. Next, we explain details and data to maximize the quality-weighted habitat objective, including monthly habitat suitability classification using streamflow, gradient, water temperature and geomorphic condition habitat criteria for each reach and how habitat connectivity is represented in the model. Then, we describe barrier passability ratings and barrier removal costs. Finally, we summarize the seasonal economic water demand functions that minimize water scarcity costs from removing water supply barriers (Figure 2). We end this section with a description of model runs.

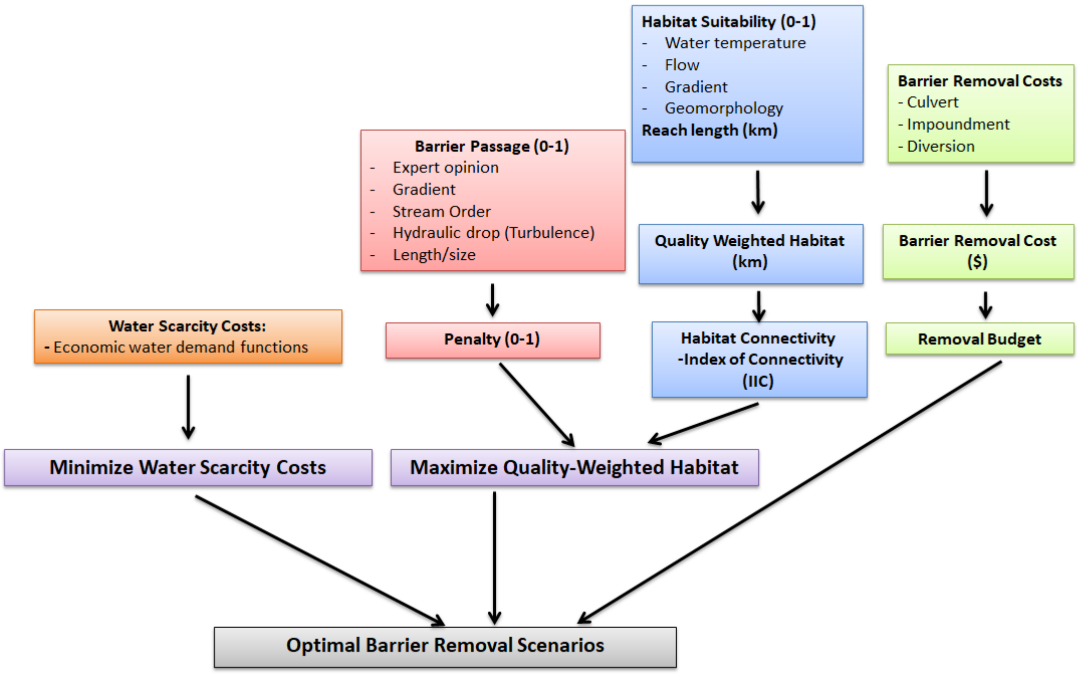

FIGURE 2. Inputs to the dual-objective optimization model that maximizes quality-weighted habitat and minimizes water scarcity costs subject to a removal budget.

Model Formulation

A linear optimization model maximized quality-weighted fish habitat (km/month) and minimized water scarcity costs for urban water uses (US$/month), constrained by a removal budget (US$/month). The model was developed in the General Algebraic Modeling System (GAMS, 2013). Some decision variables, like removing barriers and reconnecting stream reaches, were binary.

The first objective maximized connected, quality-weighted habitat between barriers i and j (Equation 1). The second objective minimized water scarcity costs resulting from lost water deliveries to urban users when a barrier is removed (Equation

2). The model does not explicitly represent time, rather time is defined by input into the model.

$$\text{Maximize: Zhabitat} = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} \frac{H_i * H_j}{1+L_{ij}} * CR_{ij} * P_j * P_i + \sum_i H_i^2}{H_L^2}, \quad i \neq j \tag{1}$$

$$\text{Minimize: Zscarcity} = \sum_k \frac{c_k}{\max(c_k)} * B_k \tag{2}$$

Where, H_i and H_j are the unimpeded distance (km/month) of quality-weighted habitat above barriers i and j (Figure 3). L_{ij} is the topological distance between the two barriers (unitless), CR_{ij} is the binary decision of reconnecting habitat between barriers i and j by removing intermediary barriers {0,1}. H_L is total quality-weighted habitat in the watershed (km/month), and P_i and P_j are passability penalties ($0.1 \leq P_i$ or $P_j \leq 1$) on barriers i and j, where values of 1 are impassable barriers and 0.1 are completely passable. Passable barriers were rated as 0.1, rather than 0, to avoid excluding passable barriers from barrier removal decision-making. In Equation 2, c_k represents the water scarcity costs ($/month) from removing barrier k and B_k is the binary decision to remove barrier k from the stream network {0, 1}.

We combined the two objective functions (equations 1 and 2) into a single objective optimization problem using the weighted sum method by applying weights on each objective which sum to 1 (Equation 3) (Cohon and Marks, 1975). The quality-weighted habitat objective, Zhabitat, ranged between 0 - 1, while water scarcity losses (c_k, equation 2) were normalized between 0 - 1 when combining the objectives into a single function. Data for economic water scarcity costs and quality-weighted connected habitat is month specific. Here, we focus model implementation on August conditions when we expect water temperature and streamflow most limit Bonneville cutthroat trout habitat and populations (Carlson and Rahel, 2010; Young, 2011), water scarcity costs are highest, and competition exists between quality-weighted connected habitat and urban water deliveries.

$$\text{Maximize } Z = (1-w)^* Z_{habitat} - (w * Z_{scarcity}) \tag{3}$$

where w = weight on objective ($0 \leq w \leq 1$).

Model constraints represent physical, habitat, and economic bounds. Equation 4 defines a reconnected reach as existing only when all barriers between i and j are removed. Equations 5 and 6 specify that reconnecting reaches and barrier removals are binary decisions, thus barriers are either fully removed or not removed. A removal budget limits barriers removed based on removal costs (Equation 7).

$$CR_{i,j} \leq \Sigma_k Int_{i,j,k} * B_k / \Sigma_k Int_{i,j,k} \text{ , } \forall\, i \neq j \tag{4}$$

$$CR_{i,j} \in \{0,1\}, \forall_{i,j} \tag{5}$$

$$B_k \in \{0,1\}, \forall_k \tag{6}$$

$$TC \geq \Sigma_k C_k{}^*B_{k,} \ \forall_k \tag{7}$$

where, $CR_{i,j}$ is the binary decision of reconnecting habitat between i and j by removing intermediary barriers {0,1}. The parameter, $Int_{i,j,k}$ is a binary parameter that indicates barrier k is in the reach between barriers i and j. Thus, the $CR_{i,j}$ variable takes a value of 1 when the numerator (count of removed barriers along the path) equals the denominator (count of all barriers along the path). Otherwise, $CR_{i,j}$ equals 0 (Equation 4). If no barriers are removed, the habitat upstream of barrier i, is counted as available habitat (H_i). The parameter, C_k is the cost of removing barrier B_k and TC is the barrier removal budget.

Figure 3 illustrates a simplified barrier network and decisions. If no barriers are removed, the decision to reconnect habitat between barriers A and B (CR_{AB}) is 0 and the unimpeded quality-weighted habitat upstream of barrier A (H_A) is included in the calculation of potential habitat (Figure 3a). $CR_{A,B}$ and topological distance ($L_{A,B}$) are 1 when barrier B is removed because the reach between barriers A and C was reconnected (Figure 3b).

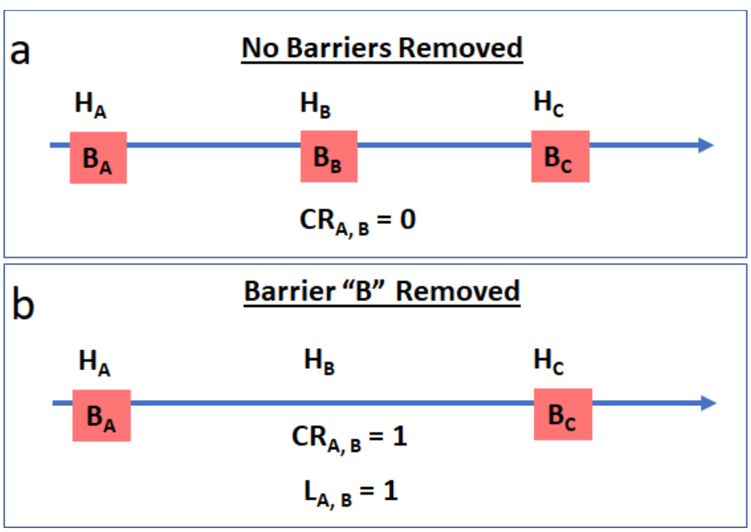

FIGURE 3. Schematic of a barrier network. When no barriers are removed (a), the decision to reconnect habitat between barriers A and C ($CR_{A,C}$) is 0. Quality-weighted habitat above barriers A and B is represented by H_{AB}. When barrier B is removed (b) a reach is created with a downstream barrier, C, and upstream barrier A. $CR_{A,B}$ is 1 and the topological distance, $L_{A,B}$, is 1.

Environmental Objective: Habitat Suitability

Habitat criteria, such as monthly percent of mean annual discharge, monthly water temperature, gradient, and geomorphic condition were intersected for each month and stream reach in a GIS database. We use habitat criteria to classify habitat suitability and, thus, quality-weighted habitat (Figure 2). The intersection classified reaches into excellent, good, fair, and poor habitat suitability (Table 1). Lindley *et al.*, (2006) and Null *et al.*, (2014) previously used a similar habitat suitability classification for steelhead trout in California streams. Merovich *et al.*, (2013) used landscape data such as elevation, geology, land cover and drainage area to predict stream conditions at multiple watershed scales in a heavily mined region of the Appalachian. Although differing in approach, numerous other studies have applied habitat classification and scoring for fish species in other watersheds (Burnett *et al.*, 2003; Quist, Rahel and Hubert, 2005; Nunn and Cowx, 2012).

TABLE 1 Habitat criteria to determine Bonneville cutthroat trout habitat suitability. All criteria must be met for excellent, good and fair habitat suitability.

	Flow October-March (% of MAD)	Flow April-September (% of MAD)	Water Temperature (°C)	Gradient (%)	Geomorphic Conditions	Rating
Excellent	> 25%	> 60%	0 - 15	0 - 6	Good or Intact	1
Good	> 12%	> 40%	0 - 18	0 - 9	Good or moderate or intact	0.75
Fair	> 5%	> 10%	0 - 21	0 -10	good or poor or moderate or intact	0.25
Poor	0 - 5%	0 - 10%	>= 21	> 10	good or poor or moderate or intact	0.10

Discharge. Average monthly discharge was extracted for each reach from the National Hydrography Plus Dataset (NHD), which has 1971-2000 gage-adjusted streamflow (U.S. Geological Survey, 2013). The NHD data set is a suite of geospatial data products including modeled streamflow using the Enhanced Runoff Method at a 30 m ground spacing resolution (McKay *et al.*, 2012). NHD estimated flow compared to 2005-2015 measured flow has a standard error of the estimate (SEE) of 2.3 m^3/s, percent bias (PBIAS) of 29.5%, R^2 of 0.96 and root mean square error (RMSE) of 2.3 m^3/s. At low flows, NHD estimated discharge nears the one-to-one line, while at high flows NHD underestimated streamflow (Kraft 2017). A modified version of the Tennant environmental flow method estimated required instream flows as a percentage of mean annual discharge (MAD) for the Weber Basin, with classifications of poor, fair, good and excellent (Orth and Maughan, 1981) (Table 1). The Tennant method is the most widely used instream flow classification method (Pyrce, 2004; Gopal, 2013) and assumes a proportion of MAD is necessary to maintain healthy ecosystems. Less than 10% of MAD is considered severely degraded fish habitat, comprising unsuitable depths, velocities and substrate. Maintaining suitable habitat for aquatic life requires flows that are at least 30% of MAD, while outstanding or optimum classification requires flows that are 60-100% of MAD (Orth and Maughan, 1981; Jowett, 1997; Gopal, 2013). Mann (2006) tested the Tennant method in the western U.S. including Utah, and found the method appropriate as a general recommendation of environmental flow, but not suitable for all regions and not representative of high gradient streams.

MAD was computed with 10 – 30 year historical flow data prior to large dam and diversion developments upstream of the gage for reaches aggregated by Strahler stream order. Reaches were grouped by stream order because historical flow data was not available for every reach in the basin. Then, October – March and April - September flow classification was calculated based on percentage of NHD average monthly flow to MAD (Table 1).

Water Temperature. Average monthly water temperature was correlated from 2005-2015 PRISM 4 km air temperatures and August 10-year average NorWeST stream temperatures (Prism Climate Group, 2016; Isaak *et al.*, 2017). Scully (2010) calculated that mean absolute error (MAE) of gridded PRISM air temperatures across the United States were 0.72 to 0.74 °C and mean bias error was -0.11 to -0.13°C. Linear regression models effectively predict water temperature from air temperature in the 0 to 20 °C range at monthly and weekly time steps because they are not spatially auto-correlated compared to daily time series (Caissie 2006). At temperatures < 0° C and > 20 °C, the slope of the curve changes from evaporative cooling and snow and ground water inputs, and the linearity assumption does not hold (Mohseni and Stefan, 1999). To account for patterns of spatial autocorrelation during relatively warm August air temperatures, modeled August stream temperatures were obtained from the NorWeST dataset. NorWeST stream temperatures report root mean square percentage error (RMSPE) of 1.07°C and MAE of 0.74 °C (Isaak et al., 2017). For all other months, stream temperatures were linearly regressed from air temperatures (Equation 8).

$$T_{i,j} = 4.2168 + 0.6259*(TA_{i,j}) \quad (8)$$

where $T_{i,j}$ represents estimated average stream temperature (°C) between barriers i and j, and $TA_{i,j}$ is PRISM 10-year average air temperature (°C) between barriers i and j. Predicted stream temperatures were validated with observed 2015 average monthly stream temperatures. The 2015 observed versus predicted water temperatures had an R^2 of 0.93, MAE of 1.28 °C, RMSE of 1.55 °C, and percent bias (PBIAS) of 2% (Kraft 2017).

Stream temperatures were categorized for Bonneville cutthroat trout as poor, fair, good or excellent. Poor water temperatures exceed 21°C and excellent water temperatures are 15 °C or colder (Table 1) (Schrank et al., 2003; Hickman and Raleigh, 1982).

Gradient. Gradient was estimated with a digital elevation model (DEM). Excellent gradients for Bonneville cutthroat trout are between 0-6%, while poor gradients are over 10% (Table 1) (Kershner, 1992; Rosenfeld, Porter and Parkinson, 2000; Hilderbrand and Kershner, 2004).

Geomorphic Condition. Stream reach geomorphic conditions range from undisturbed to severely degraded, and were developed for the Weber River by the Fluvial Habitat Center at Utah State University (Portugal *et al.*, 2016). The geomorphic assessment is a simplified version of the River Styles Framework, a tool to classify and rank river reaches by hydrology, geomorphic condition, riparian vegetation, character and recovery potential (Table 1) (Portugal *et al.*, 2016).

Habitat Suitability. Discharge, water temperature, gradient, and geomorphic condition habitat criteria were intersected for each month and stream reach to classify habitat suitability (Equation 9) (Table 1). For example, a reach with excellent Bonneville cutthroat trout habitat met all conditions of gradient < 6%, good or intact geomorphic condition, water temperature <= 15°C, and discharge > 25% of the mean annual discharge between October and March, and > 60% of mean annual discharge between April through September. A reach was categorized as poor habitat if any of the following occurred: water temperature >= 21°C, gradient > 10%, or discharge < 5% of the mean annual discharge. Ratings between 0.1 to 1 quantified habitat suitabilities so they could be input into a mathematical model. Poor habitat rating of 0.1, rather than 0, was assigned because values of 0 remove the barrier as an option from decision-making.

Habitat suitabilities were compared with habitat for known populations of Weber Basin Bonneville cutthroat trout using Fisher's exact test. Fish population estimates from Trout Unlimited provide a general idea of fish locations but are preliminary data which do not vary seasonally. The p-value of < 0.001, suggests that the habitat suitability are significant in predicting observed fish presence (Figure S1).

To determine quality-weighted habitat above each barrier for each month, the longitudinal length between barriers i and j, was calculated in GIS and multiplied by habitat suitability (Equation 10).

$$Hql_{i,j} = Q_{i,j} \cap G_{i,j} \cap T_{i,j} \cap GC_{i,j},\ \forall_{i,j} \tag{9}$$

$$H_{i,j} = Hql_{i,j} * Hl_{i,j}\ \ \forall_{i,j} \tag{10}$$

For each length of stream between barriers i and j, $Q_{i,j}$ is the monthly percent of mean annual discharge, $G_{i,j}$ is the gradient (%), $T_{i,j}$ is the monthly water temperature (°C), and $GC_{i,j}$ is the geomorphic condition (unitless). In Equation 10, the habitat suitability is $Hql_{i,j}$ (unitless), $Hl_{i,j}$ represents reach length (km), and $H_{i,j}$ denotes quality-weighted habitat (km).

Habitat Connectivity. The Integral Index of Connectivity (IIC) measures the degree of habitat connectivity at the watershed-scale, ranging from 0, unconnected, to 1, a fully connected watershed absent of barriers (Pascual-Hortal and Saura, 2006). Among the proliferation of metrics available, IIC is one of the most suitable for quantifying accessible stream habitat (Malvadkar *et al.*, 2015). The IIC represents a graph network with a set of nodes (habitat patches) and links between habitat patches. We defined river reaches as habitat patches (nodes) and barriers as links between the habitat patches. The original IIC includes all habitat patches between which fish disperse (Pascual-Hortal and Saura, 2006). We adapted the IIC by only including stream reaches without barriers between the downstream and upstream barrier. In the calculation of the IIC metric, the variable $CR_{i,j}$ identifies reaches created from removing all barriers between barriers i and j, but does not consider existing habitat above barrier i. H^2_i accounts for the quality-weighted habitat above barrier i toward the overall stream connectivity (Equation 1).

If connectivity is not included, the first objective maximized quality-weighted habitat above barrier k (H_k). The second objective did not change, and the weighted sum method combined both objective functions into a single objective (Equation 11). A sensitivity analysis of the objective function without connectivity was included.

$$\text{Maximize: Zhabitat} = \sum_{k=1}^{n} H_k * B_k * P_k \quad (11)$$

Barrier Passage

Each barrier was assigned a passage rating based on the probability of Bonneville cutthroat trout moving beyond the barrier throughout the year. Fish passage weights are from a Trout Unlimited study where potential and known barriers were categorized and passage was rated. Trout Unlimited used expert knowledge of barriers in the basin, previous studies of fish movement, water rights data, and areal imagery where indicators such as water turbulence, culvert length, evidence of vertical drop, skirt or apron size, and structure type estimated barrier passage (Trout Unlimited, 2014). Passage ratings of the identified barriers were further refined from the literature using stream gradient, stream order, culvert length and areal imagery as shown in Table 2 (Weaver, 1963; Warren and Pardew, 1998; Poplar-Jeffers *et al.*, 2009; Neeson *et al.*, 2015). Rating scores were based on previous classification systems where zero was completely passable, 0.3 was mostly passable, 0.6 was partially not passable, and 1 was not passable (Scotland & Northern Ireland Forum for Environmental Research, Edinburgh, 2010; King and O'Hanley, 2016).

A barrier was partially passable if a fish can move past the barrier only in favorable hydrologic conditions.

TABLE 2. Criteria for barrier passage classification using culvert length, water turbulence, stream order, gradient and expert opinion (Weaver, 1963; Warren and Pardew, 1998; Poplar-Jeffers *et al.*, 2009; Trout Unlimited, 2014; Neeson *et al.*, 2015).

	Rating	**Slope (reach)-GIS derived**	**Strahler order-GIS derived**	**Length of Culvert (m)**	**Box Culvert Length (m)**	**Water turbulence for all structures**
passable	0.1	< .04	> 5	<= 10	<= 100	low
mostly passable	0.3	.04 - .05	<= 4	11 - 30	100 - 400	moderate
partially not passable	0.6	>.05 -. 06	<= 4	31 - 85	>400 - 750	high
not passable	1	> .06	<= 4	> 85	>= 750	high

Barrier passage ratings were incorporated into the model as barrier penalty parameters P_i and P_j (Equation 1). Higher penalties were assigned to un-passable barriers and lower penalties to less obstructive barriers to nudge the model to remove more inhibitive barriers (Table 2).

Barrier Removal Costs

Culvert removal/replacement costs were estimated from known culvert length or measured culvert length in areal imagery. Culverts between 6.1 – 15.2 m (20 and 50 ft long), typically used for two lane roads, were estimated at $150,000 while those over 15.2 m (50 ft), typical for four lane roads, were estimated at $75,000 per lane or $300,000 (Salt Lake City Department of Public Utilities, 2008; Neeson *et al.*, 2015). Removal costs of culverts less than 6.1 m long (20 ft), were calculated from cost estimates of culvert removals in Idaho (Dupont 2000). The equation based off Dupont's (2000) estimates relate culvert length (CL) and cost of building materials, adjusted for inflation, to estimate culvert removal and bridge replacement costs (measured in $/ft) (Equation 12). The initial value of $33500 represents the base estimate for bridge replacement.

$$\text{Cost} = 33500 + 804 * \text{CL} \qquad (12)$$

Diversion removal costs were estimated from expert opinion and, if known, diverted water quantity and diversion structure size. Large diversions, primarily for municipal water use with capacity of 28.3 m^3/s or more, were estimated at a removal cost of $1 M (per comm. Paul Burnett, Trout Unlimited, 2016). Costs of small diversions, less than 28.3 m^3/s, were estimated at $300,000 (per comm. Mitigation Commission 2016).

Dam removal costs are from the American Rivers database and past large dam removal estimates in the U.S. (American Rivers, 2015). Dams with an unknown height were assigned the average cost ($250,000) of 0.3 – 1.5m (1 and 5 ft) high dam removals. Klamath Dam removal costs were compared to Weber Basin large dams height, length and reservoir capacity to estimate removal costs for dams with capacity over 1 Mm^3 (US Dept. of Interior, US Dept. of Commerce and National Marine Fisheries Service, 2012).

Six large dams in the Weber Basin were estimated to cost $30 M for removal, except the largest reservoir in the basin, Pineview Dam, which was estimated at $50 M.

Economic Objective: Water Scarcity Costs

It is important to consider economic water uses in water resources and barrier removal modeling since population in the Wasatch Front and Weber watershed continues to grow, potentially changing water demands. Managing water resources as economic goods enables resource management to mitigate water scarcity and dynamically represent water management and decision-making (Van der Zaag *et al.*, 2006). We applied estimated seasonal urban economic loss functions in the Ogden metropolitan area that were developed by Null (2018) using the demand function method. This approach requires water price (per comm. Jackson-Smith, 2018), volume of water applied at that price (Jackson-Smith, 2017), urban population (US Census Bureau, 2012) and the price elasticity of water demand (Coleman 2009). The loss functions used 2010 data for the Ogden metropolitan area, except water demand price elasticities were estimated for Salt Lake City in 1999-2003. For more detail see Null (2018).

Economic loss functions include the monthly prices that residential, commercial, industrial, and institutional water users would be willing to pay for water (Draper *et al.*, 2003; Jenkins *et al.*, 2003; Whitelaw and Macmullan, 2014). Water deliveries that meet or exceed target water demands result in no water scarcity (economic losses). When water deliveries are less than demand, water scarcity represents costs incurred to users (Jenkins *et al.*, 2003). During summer months, water demands are greater, sometimes resulting in increased water scarcity. Loss functions provide the marginal willingness to pay and scarcity cost estimates for an additional unit of water. Water demand elasticities, and thus economic loss functions, are most accurate for small changes around historical water demands and deliveries. However, most observed changes in water supply are more substantial (Ward, 2009). If water deliveries do not remain within the price range of estimated elasticity, economic losses would be underestimated. However, improving estimates would require assumptions about future demand elasticity, water price, and level of conservation, which are difficult to approximate reliably.

Monthly economic losses were estimated for seven water supply reservoirs and three major diversions. To estimate urban water scarcity losses, we assumed the 30-year average monthly flow downstream of reservoirs was equal to water demands, resulting in no water scarcity. Water scarcity costs were calculated as percent change in water delivered before and after dam removal, where 100% of water delivered resulted in zero economic loss and 5% water deliveries resulted in water scarcity losses ranging between $129 M and $856 M per month for the watershed, depending on season (Figure S2). Removing large diversions resulted in no water deliveries to the downstream demand area because we assumed that without diversions no water could be delivered.

Model Runs

We ran the model for six alternatives representing habitat suitability and water scarcity conditions in August (Table 3). The base case model was implemented at multiple barrier removal budget levels, ranging between $0/month to a budget sufficient to remove all barriers in the network, about $317.2 M/month. For each budget, the weight between the economic and environmental objectives was varied between 0 and 1 to generate alternatives along the Pareto front.

TABLE 3. Optimization model alternatives

Basecase
Without Connectivity Index
Without Barrier Passage
50% Increase to Barrier Removal Costs
25% Decrease to Barrier Removal Costs
50% Decrease to Barrier Removal Costs

We also performed extensive sensitivity analyses to explore how sensitive the modeling approach and results are to input data and assumption changes. To evaluate the sensitivity of model results to habitat connectivity, one run did not use a habitat connectivity index (Equation 11). Next, barrier passage was removed from the environmental objective function, providing results assuming that all barriers are completely impassable. Lastly, to bracket the range of results with uncertain barrier removal costs, barrier removal costs were increased by 50% and decreased by 25% and 50%.

We focused results on August habitat conditions because water scarcity costs are greatest in summer months, resulting in competition for water. This provides the most interesting results for complex water management. In reality, when barriers are removed, they are removed in all months.

RESULTS

Habitat Benefits Versus Costs

Results show Pareto optimal solutions from varying objective weights (Figure 4a) and the increase in reconnected habitat when varying the barrier removal budget (Figure 4b). When evaluating objective tradeoffs, more than 500 km of quality-weighted, connected habitat can be added in August by removing small instream barriers without affecting water supply or incurring water scarcity costs (Figure 4a). This entails removing 337 barriers, with total barrier removal costs of just over $83 M. When seven large water supply dams and 3 diversions are removed, 124 km of habitat is added but water scarcity costs exceed $660 M/month (Figure 4a).

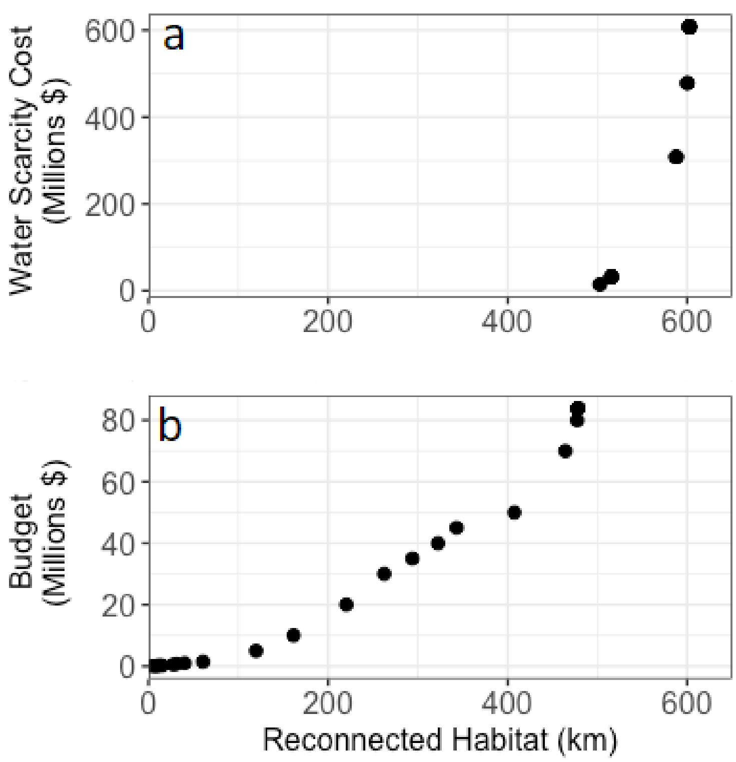

FIGURE 4. (a) Pareto optimal solutions for August habitat versus water scarcity costs and (b) tradeoff curve for August reconnected habitat versus barrier removal budget with

equal weights on both objectives. Initially, reconnected habitat costs $11,200 per kilometer, but at higher budgets increases to $1M per kilometer of reconnected habitat.

When the first two barriers are removed at a budget of $89,600, 8 km of habitat is reconnected at a cost of $11,200 per kilometer. With a budget of $10 M, 66 additional barriers are removed that connect 160 km (26%) of habitat at an average cost of $61,940 per kilometer (Figure 4b). Near a budget of $40 M, barrier removal costs increase to about $1 M per kilometer of reconnected habitat. In other words, there is decreasing marginal benefit of removing barriers, so that after the first 54 barriers are removed, costs rise to gain habitat.

Tradeoffs with Varying Objective Weights

At equal objective weights, economically important barriers are never removed in August, despite a sufficient budget. Water scarcity costs are incurred in June, July and September. At a 55% weight maximizing habitat, August water scarcity costs are $14.5 M and 502 km of habitat is reconnected ($28,900 water scarcity losses per km reconnected habitat) (Figure 4a). An additional 72 km of reconnected habitat and $276.7 M in water scarcity losses occur as the weight for maximizing habitat increases from 70% to 80% (Figure 4a).

When maximizing habitat receives 100% weight between the two model objectives and removal budget is gradually increased, water scarcity costs begin when the barrier removal budget is $10 M (Figure 5). If quality-weighted habitat is weighted by 98%, one diversion (Stoddard Diversion) is removed with a budget of $10 M (Figure 5) and is the only large barrier removed until the barrier removal budget reaches $70 M. As maximizing habitat is given smaller relative weights, water scarcity costs are incurred at higher budget levels. If 85% weight is given to maximizing quality-weighted connected habitat, again only the Stoddard Diversion is removed with a barrier removal budget of $70 M (Figure 6). At 75% weight maximizing habitat connectivity (25% minimizing water scarcity), barriers resulting in water scarcity are not removed until the budget reaches $80 M.

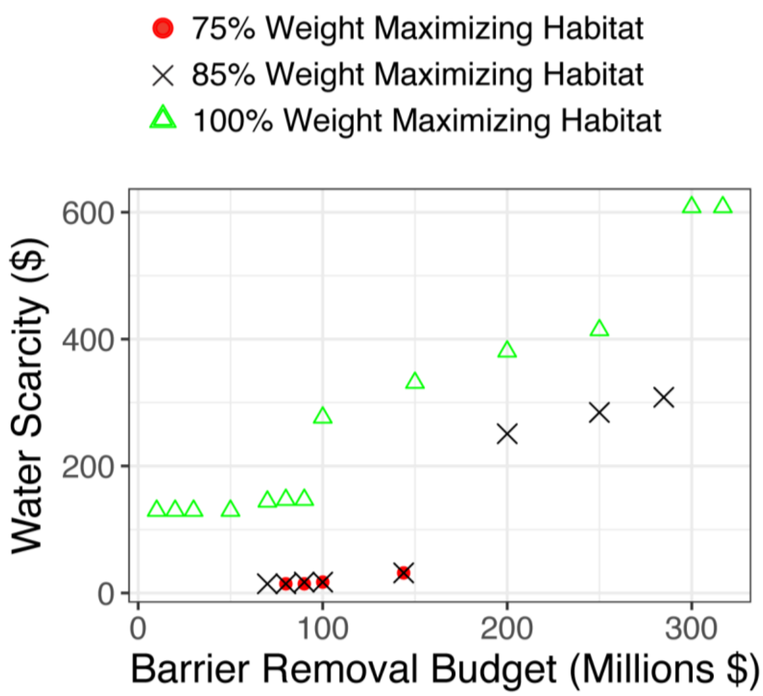

FIGURE 5. Water scarcity and barrier removal costs with varying weights between model objectives. At equal objective weights, no barriers resulting in water scarcity are removed.

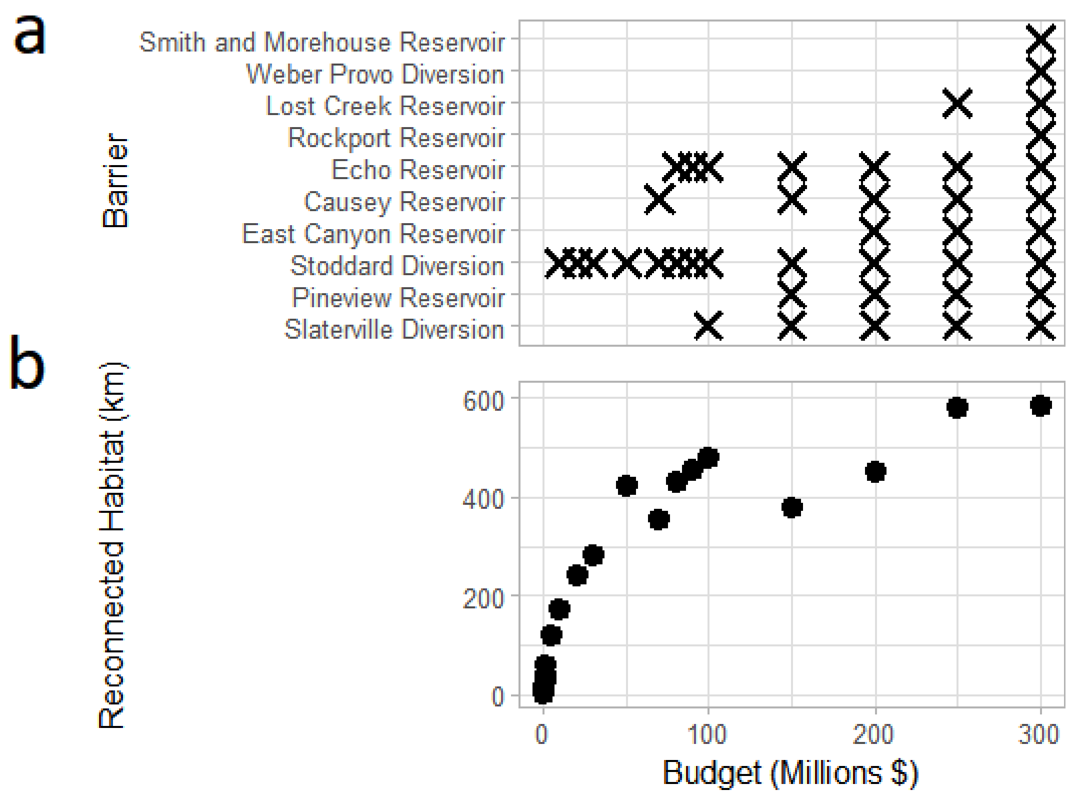

FIGURE 6. Barrier removal budgets and reconnected habitat tradeoffs when large barriers are removed. Tradeoff curve (a) and barriers removed (b) are for August habitat suitability and 85% weight on quality-weighted connected habitat.

With an $80 M budget, the longest connected reach length is 286 km when the quality-weighted objective is prioritized compared to equal weights on both objectives (Figure 7). Interestingly, with 100% weight on the maximizing habitat objective, average reach length is shortest (4 km) and the average reach length is longest (24 km) when minimizing water scarcity costs are prioritized (24 km). Average reach length is 22 km with equal weights (Figure 7).

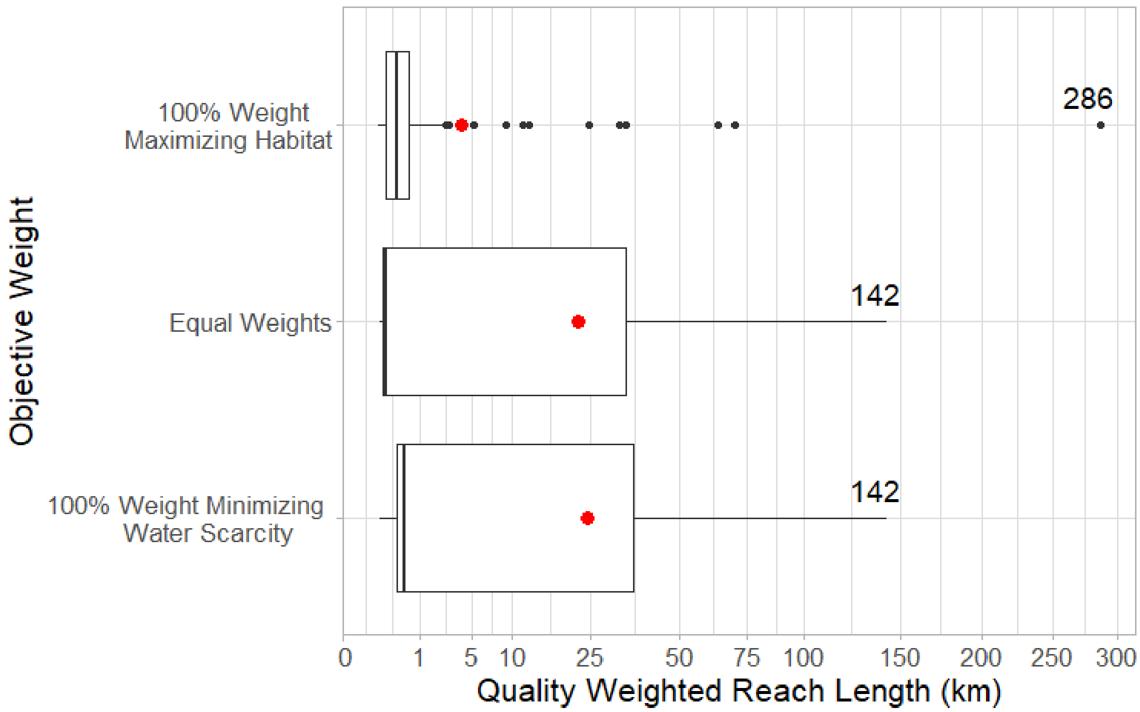

FIGURE 7. Connected reach length with different objective weights and a $80 M budget. The red dot represents the average reach length and maximum reach lengths are labeled.

Sensitivity Analyses

Including a connectivity index in model formulation allows control over the ideal length of habitat. When maximizing quality-weighted habitat without a connectivity index, the model reconnects more habitat for given barrier removal budgets until about 400 km of habitat has been reconnected (Figure 8). In August, the biggest difference occurs at 333 km reconnected habitat, where removing barriers without adding the connectivity index costs $21 M less than when the connectivity index is included. Also, without the connectivity index more habitat can be connected, but at a higher cost (Figure 8).

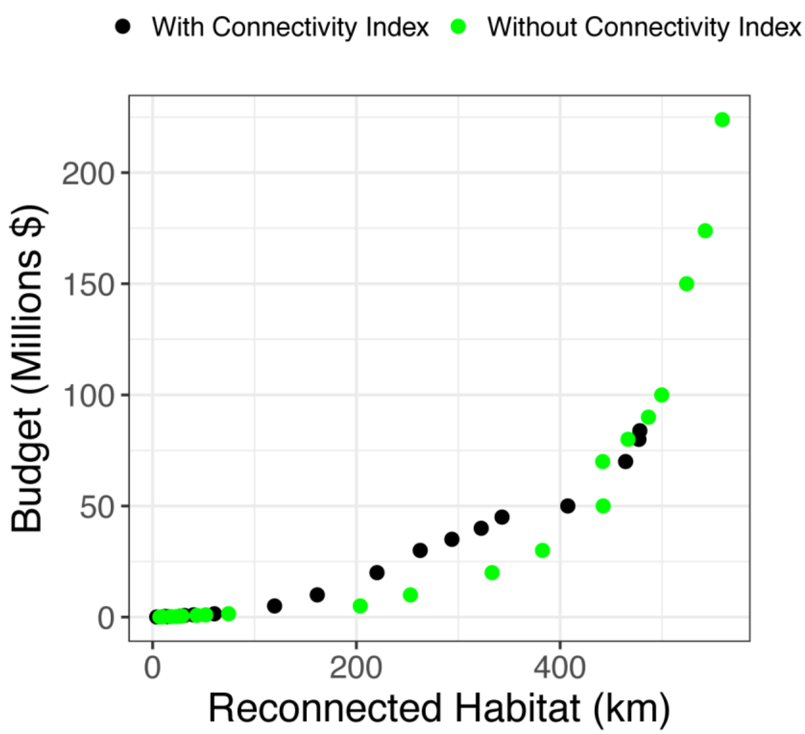

FIGURE 8. August tradeoff curve of barrier removal budget versus total habitat gain with and without a connectivity index for quality-weighted habitat.

Incorporating the probability that fish can pass barriers as a penalty in the model highlighted barriers that inhibit fish movement. When fish passage probability was not included in the model, 42% (5/12) of removed barriers were mostly or fully passable at a $1.5 M budget. When barrier passage probability was included as a penalty, 30% (3/10) of removed barriers were mostly passable and the model did not remove any fully passable barriers. While removing fully passable barriers may help restore a stream to its natural state, it may not improve fish habitat connectivity.

Barrier removal costs are uncertain, so we explored how cost changes affect results. At budgets below $10 M with 100 km of reconnected habitat, barrier removal costs do not greatly affect results (Figure 9). Between 100 km and 450 km of reconnected habitat, the marginal cost of connecting habitat increases as barrier removal costs increase, ranging between $5 M (50% cost reduction) to about $15 M (150% increase in barrier removal costs). Between 450 km and 500 km of reconnected habitat, the budget required to add additional habitat rises sharply in all cases.

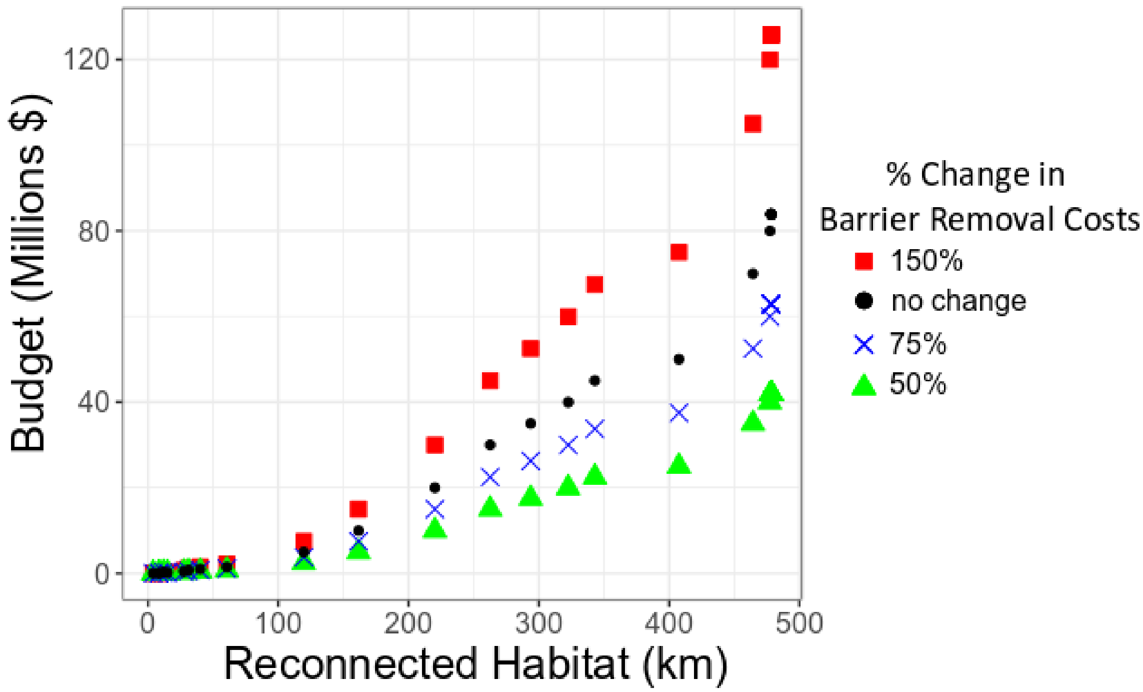

FIGURE 9. Sensitivity analysis of barrier removal costs on reconnected habitat. Barrier removal costs were increased and decreased between 150% and 50% from the original estimates.

Finally, we briefly tested results using alternative monthly input data. Differing monthly habitat suitability conditions changed water scarcity losses and barriers removed. For example, using a budget of $100 M in May, 85% weight maximizing habitat resulted in $17 M less water scarcity losses and 1 km less reconnected quality-weighted habitat than August, although both months removed 273 barriers.

DISCUSSION

Initially the marginal cost of reconnecting habitat is $11,200 per kilometer, but as the least expensive barriers are removed, marginal costs rise to $1 M per kilometer of reconnected habitat. Identifying the best river restoration investments and economic thresholds to gain the most habitat at the least cost is important for barrier removal decisions. Barrier removal cost estimates per kilometer of habitat gained are in the same range as past research on small barrier removal (Wait *et al.*, 2004; Bernhardt *et al.*, 2005; O'Hanley and Tomberlin, 2005; Reagan, 2015). For example, Wait *et al.*, (2004) reported costs ranged from $17,402 to $405,755 per kilometer of habitat in Washington streams, adjusted to 2018 dollars using an average annual inflation rate of 2.04% (Bureau of Labor Statistics, 2018).

More than 500 km, or about 80% of the quality-weighted habitat, could be reconnected by removing small instream barriers without water scarcity. The model only removes economically costly barriers after nearly all other barriers have been removed because water scarcity and removal costs are greater for large economically important barriers. Thus, focusing on small barrier removal is potentially effective to improve habitat connectivity while minimizing water scarcity costs.

A single reach was longest (286 km at $80 M budget) and average reach length shortest (4 km) with 100% weight given to maximizing the habitat objective. The average reach length was longest (24 km) with 100% weight minimizing water scarcity costs. As weights favored minimizing water scarcity costs, large, economically important barriers were not removed, creating patches of habitat (Figure 10). Rather than one single large connected reach, the model grouped barrier removals, creating numerous smaller connected reaches. If restoration goals include removing all barriers from an area, there may be a limit to maximum reach length if human water uses are also prioritized. However, focusing barrier removal in one area, rather than spreading efforts throughout the entire watershed, could improve habitat connectivity to maintain critical populations of fish (Budy *et al*., 2014). Maximizing quality-weighted habitat without including a connectivity index reconnected more habitat at a cheaper price, although habitat is spread throughout the watershed instead of centered together. Increasing quality-weighted connected habitat came as a tradeoff with cheaper, but disconnected habitats. Disconnected habitats can be important for non-migratory species and our results suggest that adjusting ideal reach lengths is promising to represent numerous or disparate species. However, if reaches remain fragmented or inaccessible, habitat gains may not benefit migratory species with large ranges, like Bonneville cutthroat trout.

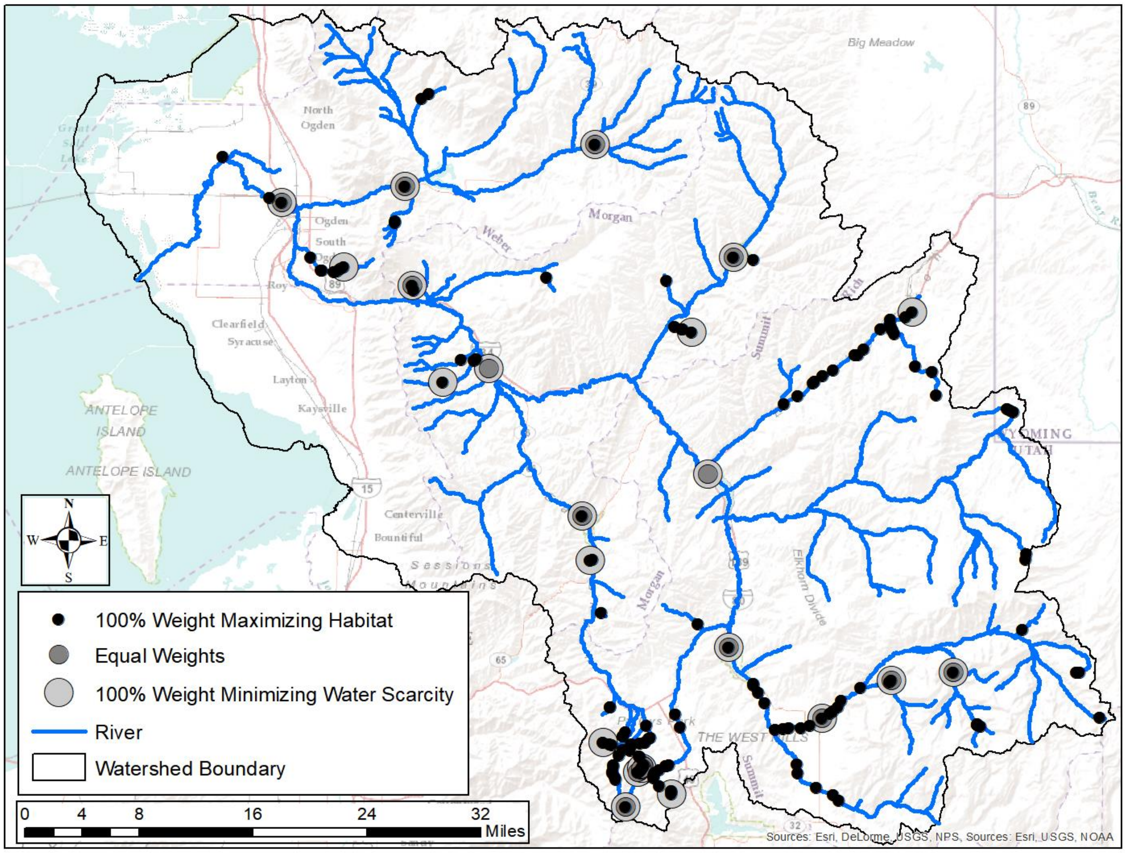

FIGURE 10. Remaining barriers with a total barrier removal budget of $80 M and 100% weight on quality-weighted connected habitat, equal weights, and 100% weight on minimizing water scarcity costs.

Regardless of objective weight, some barriers are consistently removed, indicating potential barriers that block access to quality-weighted connected habitat without water scarcity losses (Figure 10). Where circles overlap in Figure 10, barriers are consistently removed for multiple optimal solutions along the Pareto front. This highlights commonalities for managing water between competing water objectives.

Identifying Seasonally-Variable Limiting Aquatic Conditions

Although we focused mostly on August results, during different times of the year changing environmental conditions limit habitat suitability. In our model formulation, this changes which barriers are prioritized for removal, which is helpful to analyze barrier removals and make informed decisions. In reality, barriers would be removed for all months. In summer months, the primary limitation to suitable habitat is discharge and temperature, while in spring months the main limitations are gradient and geomorphic condition (Kraft, 2017). Several barriers are identified as potential candidates to be removed, depending on limiting environmental conditions at each barrier, where August habitat suitability primarily limited by water temperature, September is discharge, November is gradient and April is geomorphic condition (Figure 11).

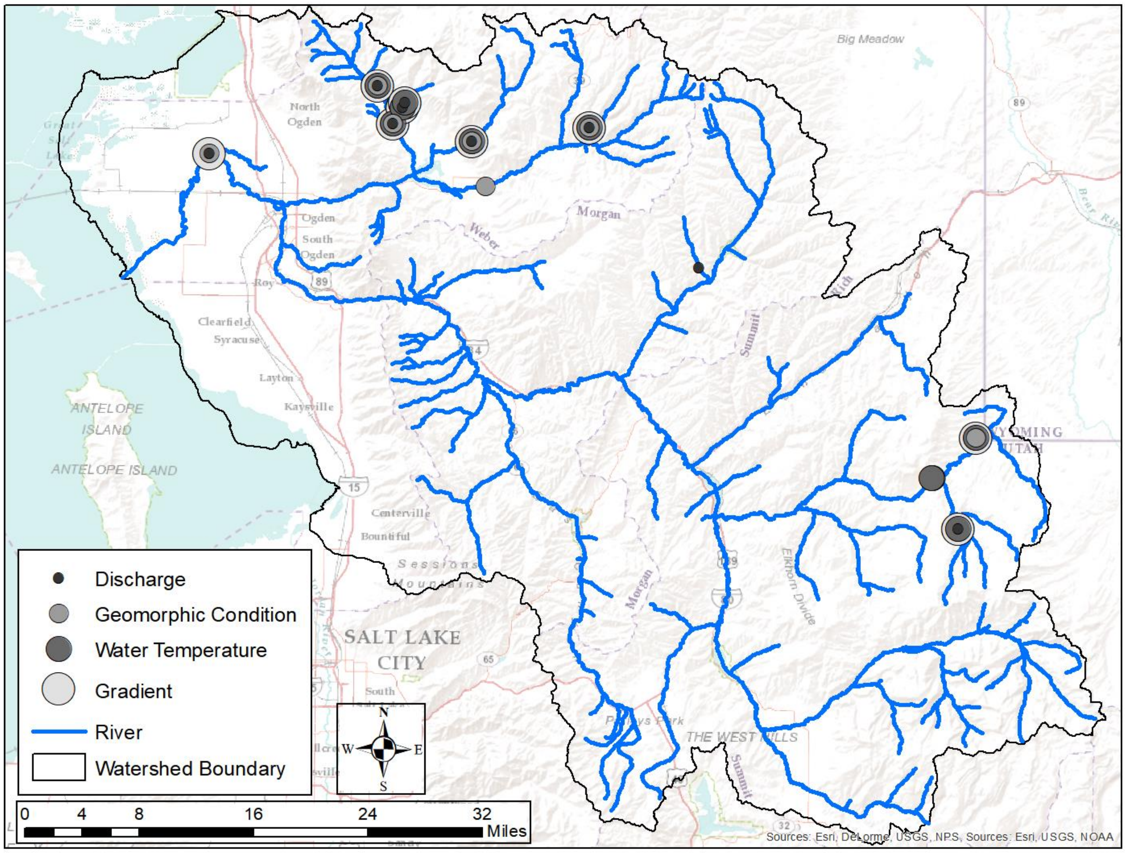

FIGURE 11. Promising barriers to remove with the inhibiting aquatic habitat condition at each barrier.

Assessing which physical and water quality attributes limit habitat is important for restoring and access to habitat for desired fish populations. To restore Bonneville cutthroat trout habitat in the Weber Basin, increasing discharge and decreasing water temperatures during summer months, and simultaneously improving access to suitable habitats could potentially restore viable populations.

Types of Barriers Removed

Diversions and road crossings were always the most frequently removed barriers. Road crossings were, on average, cheaper than other barriers, but they make up only about 24% of all barriers, so their removal recurrence suggests they play a key role in improving habitat connectivity. Small diversions were also commonly removed, likely because most instream barriers are small diversions.

LIMITATIONS

Data availability and simplifications limit modeling. We assumed that barrier passage by fish is constant throughout the year, and the same for fish moving upstream or downstream. Future work could expand barrier passage ratings and include cumulative passability. Also, considering alternative restoration options such as fish ladders in models may be promising at expensive and large barriers to reconnect fish habitat without affecting water scarcity costs. Costs of barrier removals were estimated and generalized to illustrate barrier removal options. Improving barrier removal cost estimates is a needed direction for future research. The only economic water use considered here was for urban water supply from ten barriers. And yet some of those barriers also provide hydropower, flood protection, and recreational benefits which could be added to future models.

As large barriers are removed, reach habitat quality changes. For example, removing large dams could return rivers to a natural flow regime (Poff *et al.*, 1997) and temperature regime. Cold water reservoir releases that benefit downstream fish populations may be lost (Rheinheimer *et al*., 2015). This dynamic habitat change was not accounted for as barriers were removed, which had a minor effect as most barriers prioritized for removal were small structures (Bednarek, 2001).

The model was implemented for a particular month, August, where habitat was limited, urban water demands were large, and tradeoffs between quality-weighted connected habitat and water scarcity cost objectives were most pronounced. This assumed habitat and water scarcity conditions persisted for the entire year. In reality, those conditions change and the model objective function could be extended to instead aggregate changing conditions.

The Weber Basin barrier removal model included only natural, perennial rivers. Canals, ditches and small intermittent streams were assumed not to provide suitable habitat for fish and were not included. We assumed increasing suitable habitat for Bonneville cutthroat trout would increase fish productivity. However, additional fish species and life stages could be included in future work or for other watersheds. Interannual variability of stream flows and habitat was also not considered, although monthly variability was considered. Finally, our model maximized total length of suitable habitat. Mainstem and tributary reaches were treated equally; however, reaches with tributary confluences provide diverse habitat and may be preferred ecologically over a single mainstem reach. In future work, it would be beneficial to better incorporate river topology when considering barrier removal.

SUMMARY AND CONCLUSIONS

This paper prioritized barrier removal using dual-objective optimization to maximize quality-weighted, connected habitat and minimize water scarcity costs of reduced water deliveries to cities. Our model incorporated habitat suitability from discharge, water temperature, gradient and geomorphic condition. A habitat connectivity index estimated each barrier's contribution to habitat connectivity. Ability of Bonneville cutthroat trout to move beyond a barrier was represented by barrier passability penalties, where impassable barriers received a greater penalty and thus were more likely to be selected for removal. Economic losses due to lost water deliveries were considered for seven reservoirs and three diversions. A budget for barrier removal constrained the model. Results were visualized as a Pareto-optimal tradeoff curve, where each point on the curve represented a different set of barriers to be removed. Tradeoff curves of habitat gain versus water scarcity costs and barrier removal costs visualized results for decisions makers to evaluate.

Five main conclusions illustrate the advantages of barrier removal optimization modeling, using our results from the Weber Basin. First, there are diminishing returns to river restoration investments for connected habitat as more barriers are removed. The initial $10 M spent on removing barriers connected more suitable habitat per dollar than the last $10 M. Understanding habitat gains over a range of barrier removal restoration budgets is beneficial for watershed managers to make restoration decisions.

Second, removing numerous small barriers connected more habitat with lower water scarcity costs from lost water deliveries, compared to removing large, water supply barriers. Removing large barriers was expensive and resulted in less cumulative habitat gained. Road crossings were the most frequent barriers chosen for removal, indicating they currently fragment suitable habitat in the Weber Basin and removing or retrofitting them is promising for restoration.

Third, water scarcity costs are important to consider as a model objective. When only aquatic habitat was maximized, water scarcity losses began at a barrier removal budget of $10 M and were greater than the dual-objective model at all budget levels.

Fourth, model results change depending on management preferences and questions. When habitat suitability was optimized without a connectivity index, connected habitat was patchy, and was often inaccessible for migratory species. The ability to adjust the model inputs, habitat coefficients and analyses allows flexibility to apply barrier optimization to different watershed networks and fish species. For example, changing input habitat suitability criteria for another fish species produces a different set of results. Instead of focusing on August habitat conditions, it may be more suitable to identify barrier removal projects benefiting habitat conditions during a different season. Similarly, keeping some barriers in place (excluding the barrier as a removal option) could be a tool for decision-makers to block the spread of invasive species.

Fifth, optimization modeling is a promising approach to consider both human (economic) and environmental objectives in river restoration and water resources management. Our optimization model successfully incorporated numerous objectives and habitat criteria to determine promising restoration solutions given human water needs.

Overall, tradeoffs exist between quality-weighted aquatic habitat connectivity and water scarcity costs. However, removing numerous small barriers did not affect water supply or incur water scarcity costs at budget levels below $10 M, connecting quality-weighted habitat at the least cost, compared to removing large dams and diversions. If an economically important barrier is detrimental to aquatic habitat, understanding the barrier’s economic importance and potential improvement to aquatic habitat is needed prior to decision-making. It was never optimal to remove water supply dams or diversions even when aquatic habitat was prioritized over water supply.

Water supply has historically been prioritized in arid, semi-arid, and Mediterranean climates. However, large-scale reductions in habitat, species, ecosystem services, and water quality have led to recent notable instances where water supply infrastructure was removed or re-operated to enable habitat restoration, such as dam removals on the Snake and Elwha Rivers (Kruse *et al.*, 2006; US Dept. of Interior, US Dept. of Commerce and National Marine Fisheries Service, 2012). Our model results and utility were communicated with local watershed managers and decision-makers. Our model quickly prioritized barrier removals that are currently being considered by water managers in the Weber Basin, such as the Pacificorp Weber Dam and the Stoddard Diversion, where fish passageways are being added (Trout Unlimited and UDWR, pers.comm.). Our research lends scientific credibility to restoration decision-making, and the overlap of barriers identified for removal or retrofitting by watershed decision-makers corroborates our model results.

This modeling approach was demonstrated with a case study in the Weber River watershed, although the optimization model is generalizable to other systems by changing input data. Removal decisions are complex when considering multiple objectives with constraints for hundreds of barriers. Optimization offers a feasible method to consider multiple objectives of connecting habitat and maintaining water deliveries at the watershed-scale. The dual-objective optimization model developed may improve decision-making for complex multi-objective problems for which decisions are not easily reversed. This work underscores the utility of barrier removal optimization for decision-making and quantifies habitat and economic effects of barrier removal, while visualizing results for watershed managers.

SUPPORTING INFORMATION

Additional supporting information may be found online under the Supporting Information tab for this article: Figure S1: Habitat suitability versus known populations of Bonneville cutthroat trout; Figure S2: Seasonal economic loss functions for the Ogden metropolitan area

DATA AVAILABILITY

Data are openly shared at hydroshare.com (Kraft and Null, 2017), and our model is publicly available on GitHub (https://github.com/MaggiK/Optimizing-Stream-Barrier-Removal).

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