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Optimizing Barrier Removal to Restore Connectivity in Utah’s Weber Basin

Maggi Kraft
Utah State University

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OPTIMIZING BARRIER REMOVAL TO RESTORE CONNECTIVITY IN UTAH’S WEBER BASIN

by

Maggi Kraft

A thesis submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

in

Watershed Hydrology

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UTAH STATE UNIVERSITY
Logan, Utah

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ABSTRACT

Optimizing Barrier Removal to Restore Connectivity in Utah’s Weber Basin

by

Maggi Kraft, Master of Science
Utah State University, 2017

Major Professor: Dr. Sarah Null
Department: Watershed Sciences

In-stream barriers, such as dams, culverts and diversions alter hydrologic processes and aquatic habitat. Removing uneconomical and aging in-stream barriers to improve stream habitat is increasingly used in river restoration. Previous barrier removal projects focused on score-and-rank techniques, ignoring cumulative change and spatial structure of barrier networks. Likewise, most water supply models prioritize either human water uses or aquatic habitat, failing to incorporate both human and environmental water use benefits. In this study, a dual objective optimization model prioritized removing in-stream barriers to maximize aquatic habitat connectivity for trout, using streamflow, temperature, channel gradient, and geomorphic condition as indicators of aquatic habitat suitability. Water scarcity costs are minimized using agricultural and urban economic penalty functions, and a budget constraint monetizes costs of removing small barriers like culverts and diversions. The optimization model is applied to a case study in Utah’s Weber River Basin to prioritize removing barriers most beneficial to aquatic habitat
connectivity for Bonneville cutthroat trout, while maintaining human water uses.

Solutions to the dual objective problem quantify and graphically show tradeoffs between connected quality-weighted habitat for Bonneville cutthroat trout and economic water uses. Removing 54 in-stream barriers reconnects about 160 km of quality-weighted habitat and costs approximately $10 M, after which point the cost effectiveness of removing barriers to connect river habitat decreases. The set of barriers prioritized for removal varied monthly depending on limiting habitat conditions for Bonneville cutthroat trout. This research helps prioritize barrier removals and future restoration project decisions within the Weber Basin. The modeling approach expands current barrier removal optimization methods by explicitly including both economic and environmental water uses. The model is generalizable to other basins by changing input data.

(103 pages)
Optimizing Barrier Removal to Restore Connectivity in Utah’s Weber Basin

Maggi Kraft

River barriers, such as dams, culverts and diversions are important for water conveyance, but disrupt river ecosystems and hydrologic processes. River barrier removal is increasingly used to restore and improve river habitat and connectivity. Most past barrier removal projects prioritized individual barriers using score-and-rank techniques, neglecting the spatial structure and cumulative change from multiple barrier removals. Similarly, most water demand models satisfy human water uses or, only prioritize aquatic habitat, failing to include both human and environmental water use benefits. In this study, a dual objective optimization model identified in-stream barriers that impede quality-weighted aquatic habitat connectivity for Bonneville cutthroat trout. Monthly streamflow, stream temperature, channel gradient and geomorphic condition were indicators of aquatic habitat suitability. Solutions to the dual objective problem quantify and graphically present tradeoffs between quality-weighted habitat connectivity and economic water demands. The optimization model is generalizable to other watersheds, but it was applied as a case study in Utah’s Weber Basin to prioritize removal of environmentally-harmful barriers, while maintaining human water uses.

Modeled results suggest tradeoffs between economic costs of removing barriers and quality-weighted habitat gains. Removing 54 in-stream barriers increases quality-weighted habitat by about 160 km and costs approximately $10 M, after which point the
cost effectiveness of removing barriers to connect river habitat slows. In other words, there is decreasing benefit of removing barriers, so that after removing the first 54 barriers, it costs more to connect more high-quality habitat. Removing reservoirs or diversions that result in large economic losses did not substantially increase habitat. This suggests that removing numerous small barriers results in greater increases in habitat for the same removal costs, without significant water scarcity losses. The set of barriers prioritized for removal varied monthly depending on limiting habitat conditions for Bonneville cutthroat trout. The common barriers removed in the model were identified to communicate the most environmentally harmful barriers to local stakeholders and inform decision-making. Additionally, limiting the budget or number of barrier removal projects resulted in a different set of barriers removed. This research helps prioritize barrier removals and future restoration decisions in the Weber Basin although the model formulation is generalizable to other watersheds. Available data and a simplified approach limit the scope of this model. The modeling approach expands current barrier removal optimization methods by explicitly including economic and environmental water uses.
ACKNOWLEDGMENTS

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CHAPTER I

INTRODUCTION

Dams, culverts and diversions, collectively referred to as in-stream barriers, are economically-important for water supply and conveyance, but negatively affect river ecosystems and disrupt hydrologic processes. In-stream barriers change the 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 in-stream barriers to improve aquatic habitat connectivity is increasingly used to restore river habitat (Stanley & Doyle 2003; Magilligan et al. 2016). Including aquatic habitat suitability and barrier passage is necessary to effectively improve environmental objectives when prioritizing barrier removal. Improving techniques to include both human water demands and aquatic habitat objectives is needed to advance understanding of environmental-economic tradeoffs to restore suitable habitat connectivity while managing competing human water uses.

Most past barrier removal research focused on identifying individual barriers to remove using a score-and-rank technique, which ignores cumulative change from multiple, spatially-connected barrier removals (O’Hanley 2011). A score-and-rank approach scores physical, economic or ecological attributes of barriers, then ranks them for potential removal. Score-and-rank is straightforward and simple, but does not consider dynamic change or cumulative effects of removing multiple barriers within the stream system (Kemp & O’Hanley 2010; O’Hanley & Tomberlin 2005). For example, when prioritizing multiple barrier removals, a score-and-rank procedure will prioritize
barriers in a listed order ignoring the spatial relationship between the two barriers. Similarly, most water supply models optimize either human water use or, occasionally, aquatic connectivity, failing to holistically represent both human and environmental benefits (Null et al. 2014). To overcome the shortcoming of score-and-rank techniques, I used dual objective optimization modeling to evaluate barrier removal given human and environmental objectives, and account for the interconnected, spatial structure of a multi-barrier network.

Optimization mathematically maximizes or minimizes specific objectives, resulting in a Pareto-frontier tradeoff curve, where points on this curve are efficient solutions for each objective (Kyle McKay et al. 2016). Two early studies to optimize barrier removal were Kuby et al. (2005) and O’Hanley and Tomberlin (2005). Kuby et al. (2005) developed a multi-objective optimization model to compare economic-ecological tradeoffs of dam removal in the Willamette River Basin in Oregon. Their model quantifies and visualizes trade-offs between salmonid migration, hydropower and water storage loss but does not include barrier removal costs. O’Hanley and Tomberlin (2005) proposed a general nonlinear optimization model to improve fish passage barrier removal from multiple barrier modification options. Later, King and O’Hanley (2016) reformatted the problem to optimize barrier passage alternatives given an implementation budget.

Structurally similar to Kuby et al. (2005), O’Hanley (2011) maximized connectivity of a single section of river given a removal budget, to improve environmental conditions of the river network. This approach was well suited for
potamodromous fish species by connecting the largest river reach from the farthest downstream barrier.

Null and Lund (2012) and Raegan (2015) included multiple options for barrier removal or restoration to restore connectivity. Null and Lund (2012) maximized fish production constrained by the cost of habitat improvement alternatives, such as increasing flow, riparian vegetation and removing a dam. More recently, Reagan (2015) developed an optimization model for culvert replacements while including replacement costs, culvert passability, and climate change scenarios. Of these barrier removal studies, none considered economic water scarcity costs in conjunction with aquatic habitat gain.

Conversely, Null, SE. (2016), Null et al. (2014) and Null and Lund (2006) used a hydroeconomic optimization model to evaluate dam removal that included economic scarcity costs of water losses in California. Null et al. (2014) minimized water scarcity of large dam removal with historical and future climate conditions. Model tradeoffs were evaluated between economic scarcity costs of dam removal with environmental benefits of improved access to suitable upstream habitat. Although aquatic habitat was included in the analysis, it was not included directly in the optimization model. Null & Lund (2006) modeled water scarcity costs in the Hetch Hetchy System with and without O’Shaughnessy Dam and Null (2016) evaluated improving water conveyance to maintain water reliability with and without O’Shaughnessy Dam. The optimization model incorporated economic water benefits to agriculture and urban water users, but it did not include aquatic habitat or environmental benefits (Draper et al. 2013).

Zheng et al. (2009) developed a multi-objective optimization consisting of nine objectives to understand tradeoffs between criteria of ecological health for multiple
species, dam removal and invasive species control costs in watersheds of Lake Erie. Economic costs were included as a function of dam removal and sea lamprey control costs. Zheng & Hobbs (2013) extended the model developed by Zheng et al. (2009) to incorporate tradeoffs between public safety and the other nine criteria. 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 temporally to understand the significance of allocating funding for restoration projects through time.

Previous barrier removal systems modeling optimized aquatic habitat connectivity, but excluded economic benefits of dams, like water supply or hydropower benefits. When costs are included, they are for dam removal or remediation (Zheng & Hobbs 2013; King & O’Hanley 2016; Reagan 2015). Similarly, water resources management systems models explicitly include economic objectives, but represent environmental criteria as constraints, removing them from decision-making (Draper et al. 2004). Some studies represented in-stream habitat overly-simplistically, as accessible drainage area or river miles (Neeson et al. 2015; Kuby et al. 2005) or did not consider passability of barriers at different flows or for different species or fish life stages (Kuby et al. 2005; Null et al. 2014; King & O’Hanley 2016).

I developed an optimization model to identify the in-stream barriers to remove that maximize aquatic habitat connectivity and minimize economic water scarcity costs. The environmental objective maximizes aquatic habitat connectivity for trout, using streamflow, water temperature, channel gradient and geomorphic condition as indicators of aquatic habitat suitability (Hilderbrand & Kershner 2004). The Integral Index of
Connectivity (Saura & Pascual-Hortal 2007) calculates the set of barrier removals contribution to improving connectivity between quality-weighted habitat. The economic objective minimizes water scarcity costs using agricultural and urban economic penalty functions (Draper et al. 2003). A removal budget constrains costs and limits the number of barriers to remove (Januchowski-Hartley et al. 2013; Null & Lund 2012). My approach is novel because it incorporates numerous variables to effectively model human water uses and quality-weighted fish habitat connectivity as dual objectives to prioritize barrier removal and inform water resources management. My optimization model is applied to Utah’s Weber Basin to prioritize the most environmentally harmful barriers, while considering human water uses; however, the model formulation is generalizable to other basins by changing input data. My case study focuses on restoring connected habitat for protected Bonneville cutthroat trout as an indicator of high quality, connected aquatic habitat.
CHAPTER II
BACKGROUND

2.1 Study Site

Utah’s Weber Basin is 6.4 Gm², spanning the high Uintah Mountains to the Great Salt Lake (Figure 1). Snowmelt from the Wasatch and Uintah Mountains is the primary source of water. The basin has a montane to semi-arid environment, receiving about 25.4 mm precipitation a year (SWCA 2014). The Weber River is highly regulated (Figure 2), averaging about 12.46 m³ (440 cfs) near the outlet to the Great Salt Lake, although stream flow would be considerably higher without consumptive water uses (Weber River Near Gateway USGS Gage, Wurtsbaugh et al. 2015).

FIGURE 1. Weber Basin in northern Utah. Dots represent small barriers such as diversion dams, impoundments, and road crossings. Large dams are represented by triangles. Known in-stream barriers are from NHD, USGS, and Trout Unlimited datasets and were combined to develop a barrier database.
Native American tribes first inhabited the region, and fur trappers and explorers lived near the Great Salt Lake in the early 1800s. The arrival of members of the Church of Jesus Christ of Latter-day Saints (Mormons) in 1847 marked the first large-scale settlement of Anglo-Americans in the region. The semi-arid environment of the Salt Lake region led the Mormons to manage and develop water resources through dams, canals and water use regulations (McCune 2000). In 1852, the Utah territory created regional water rights giving preference to crop and agricultural irrigation. In 1896, construction of East Canyon Dam was the first major dam in the Weber River Basin. Water development in the Ogden area progressed with growing population and industrialization. In 1902, the Reclamation Act marked the beginning of Federal Government control and assistance in water infrastructure. East Canyon Reservoir was expanded in 1916 to accommodate growth and increasing agricultural water demand in the Ogden Valley. The Weber River Project was completed in 1931, overseeing the construction of Echo Reservoir and the Weber-Provo diversion canal. In 1949, the Weber Basin Project was enacted to facilitate
water development and water resource use in the Weber Basin. By 1987, there were seven major reservoirs (with water storage capacity exceeding 9.25 Mm$^3$ (7500 acre-feet [AF])), two on the mainstem Weber River and three diversions supplying water to the Wasatch Front (Figure 1) (McCune 2000).

Currently, Weber River watershed supplies about 98.2 Mm$^3$ (79,600 AF) of water to municipal and industrial water users per year and 266.4 Mm$^3$ (216,000 AF) annually for irrigation (Weber Basin Water Conservancy, 2010). The basin supplies water for over 500,000 people in the Wasatch Front corridor. The Wasatch front population is projected to nearly double by 2050 to over one million people (Harbeke et al. 2014).

The Weber River historically supported healthy populations of Bonneville cutthroat trout (*Oncorhynchus clarki Utah*). Altered environmental conditions, reduced

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<th>Name</th>
<th>Construction Date</th>
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<tr>
<td>East Canyon Dam</td>
<td>1896, expanded in 1916 and 1967</td>
<td>63.2 Mm$^3$</td>
</tr>
<tr>
<td>Smith and Morehouse Dam</td>
<td>1925, expanded in 1987</td>
<td>1.03 Mm$^3$</td>
</tr>
<tr>
<td>Echo Dam</td>
<td>1931</td>
<td>91.9 Mm$^3$</td>
</tr>
<tr>
<td>Pineview Dam</td>
<td>1937</td>
<td>135.9 Mm$^3$</td>
</tr>
<tr>
<td>Wanship Dam</td>
<td>1957</td>
<td>75.6 Mm$^3$</td>
</tr>
<tr>
<td>Lost Creek Dam</td>
<td>1966</td>
<td>27.8 Mm$^3$</td>
</tr>
<tr>
<td>Causey Dam</td>
<td>1966</td>
<td>970,750 m$^3$</td>
</tr>
<tr>
<td>Weber-Provo Diversion</td>
<td>1931, expanded in 1947</td>
<td>28.3 m$^3$/s</td>
</tr>
<tr>
<td>Stoddard Diversion</td>
<td>1965</td>
<td>169.9 m$^3$/s</td>
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<tr>
<td>Slaterville Diversion</td>
<td>1969</td>
<td>254.9 m$^3$/s</td>
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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; Lentsch et al. 1997; Lentsch et al. 2000; UDWR 2009). Bonneville cutthroat trout are protected under a multi-state conservation agreement to conserve and eliminate threats to ensure long-term survival of populations and avoid listing under the Endangered Species Act (Webber et al. 2012; Lentsch et al. 2000). Considering the conservation goal of these species, restoring connectivity to provide access to suitable habitat is essential to sustain and enhance viable Bonneville cutthroat trout populations.

2.2 Environmental Consequences of In-Stream Barriers

Regulated flows and water storage from dams benefit cities and agriculture to provide reliable water supply during dry seasons and droughts in snowmelt driven streams. However, dams alter natural flow regimes. Native fish consistently respond negatively to decreased magnitude and frequency of flood events and increased base flows (Poff & Zimmerman 2010). Changes in seasonal flood timing and decreased peak magnitude disrupt important life cycle stages in fish including spawning cues for native fish species (Poff et al. 1997; Gido et al. 2013). Altered flow regimes also affect sediment transport and water quality. For example, without high spring runoff flows, sediment is not mobilized but rather is retained behind dams, changing downstream habitat structure and nutrient availability (Stanley & Doyle 2003; Poff & Hart 2002; Bednarek 2001; Petts 2009). Additionally, reduced flooding to wetlands can result in successional changes to vegetation and aquatic biodiversity (Kingsford 2000). The development of Weber Basin water conveyance and dam projects reduced peak flows,
altering hydrologic conditions (Figure 2). The stream gage plotted in Figure 2 is located farthest downstream below all major diversions and dams. Between 1957 and 1980, peak discharge and peak flow decreased likely due to the completion of the large dams and diversions, but prior to expansion of Smith and Morehouse Reservoir. However, the downstream reservoirs and barriers likely diminished the hydrologic affect of the 7300 AF expansion of Smith and Morehouse Reservoir. Small barrier removal potentially helps restore or mitigate hydrologic effects of water development on aquatic habitat (Kauffman et al. 1997).

Additionally, topography, climate, discharge and streambed characteristics control stream temperatures (Caissie 2006). When dams and diversions reduce the volume of water or thermal mass in the stream, atmospheric heating can increase water temperatures, especially during summer (Bartholow 1991; Sinokrot & Gulliver 2000). Bottom release dams discharge cool water from depth in the reservoir, and top release dams release warm water from the reservoir surface, both of which change downstream thermal regimes (Lessard & Hayes 2003; Olden & Naiman 2010). During summer months, air and stream temperatures reach their maximum, compounding temperature increases from top-release dams and reduced streamflows from diversions (Lessard & Hayes 2003). Increased stream temperatures alter suitable aquatic habitat and affect bioenergetics and assemblage composition of fish and macroinvertebrates. Removing environmentally harmful barriers may improve temperature regimes, water quality and habitat conditions.
2.3 Longitudinal Habitat Connectivity

Fragmentation occurs when habitats become separated into multiple patches, potentially reducing organism movement and total habitat area (Wilcove et al. 1986). Smaller barriers like diversion dams, weirs and culverts fragment habitats, inhibit species’ migration and movement by preventing connectivity to spawning environments, and can reduce genetic variability between populations (Peterson et al. 2013; Compton et al. 2008; Pringle 1997). Movement patterns of cutthroat trout are greatest in spring, moving distances up to 82 km per season (Schrank & Rahel 2004; Carlson & Rahel 2010; Colyer et al. 2005), although the majority of fish relocate less than 10 km within the river (Young 2011; Colyer et al. 2005). Summer and winter movement is limited to within 1 km but, at times, cutthroat trout move 21.5 km (Carlson & Rahel 2010; Colyer et al. 2005). In-stream barriers reduce the habitat available for fish to migrate. Besides inhibiting movement, diversion dams potentially entrain fish in canals that do not contain sufficient habitat or water flow (Carlson & Rahel 2010). Disconnected populations become isolated, increasing potential extinction risk (Hilderbrand & Kershner 2004).

Habitat fragmentation between metapopulations in the Weber Basin limits population dispersal and prevents access to preferred spawning reaches and other suitable habitat (Budy & Thiede 2014). Dispersal between metapopulations is important for access to higher quality habitat or preferred habitat at different life stages and maintain healthy subpopulations. Connectivity between habitats is not only important for access to suitable habitat, also maintaining genetic variation and exchange between populations (Budy et al. 2007; Budy & Thiede 2014; Pringle 1997). Disconnected subpopulations become isolated, increasing potential extinction risk (Rieman & Dunham 2000;
Hilderbrand & Kershner 2004) therefore, connectivity between and within subpopulations is important for preserving a healthy widespread population. Habitat fragmentation and access to spawning habitat remains important when considering all structures of fish populations, not only metapopulations of trout. Habitat fragmentation becomes increasingly important when considering different life cycles and migration needs for a single large population of fish.

2.4 Habitat Suitability

Cutthroat trout prefer clear, cold water and complex habitats with sufficient depth for migration, depending on life stage (Budy et al. 2007; Colyer et al. 2005; Kershner 1992; Lentsch et al. 1997). Annual spawning for Bonneville cutthroat trout usually occurs in spring and into summer at higher elevations (Bennett et al. 2014; Budy et al. 2012). Trout prefer water temperatures under 15°C (Bear et al. 2007; Cade 1985) but are able to survive in temperatures over 22°C and potentially up to 26°C for short periods of time (Schrank et al. 2003; Cade 1985). Ideal water depth for adult cutthroat trout ranges between 0.4 and 0.7 m and 0.3-0.6 m for juveniles in low velocity or gradient streams (Braithwaite 2011; Cade 1985; Kershner 1992; Rosenfeld et al. 2000). Measurements in Nebraska, Wyoming and Montana found suitable depths with stream flows of more than 30% of historic flows (Jowett 1997; Gopal 2013).

Weber Basin stakeholders have previously considered and implemented re-connecting fish habitat as part of river restoration. In 2012 the National Fish Habitat Association listed Weber River as "Water to Watch" because of their efforts to reconnect 17.5 miles (12.07 km) of fish habitat. The project, carried out through the Western Native

Given the scope and magnitude of barrier effects on river habitat and aquatic ecosystem health, removing barriers offers an opportunity to restore reaches of aquatic habitat within a watershed (Magilligan et al. 2016; Stanley & Doyle 2003). However, the number of barriers and restoration options, as well as competing water management objectives, makes it challenging to identify which barriers to remove, ultimately hindering decision-making. To restore river connectivity, it is important to understand multi-scale dynamics of barrier removal problems (Magilligan et al. 2016; Grant & Lewis 2015; Milt et al. 2017).
CHAPTER III

METHODS

3.1 Model Description

I developed a binary linear program optimization model to maximize connectivity between quality-weighted, in-stream habitat (km), and minimize economic water losses ($) for each month. My optimization model was implemented in General Algebraic Modeling System (GAMS) software. Figure 3 is an example of a stream network where red boxes represent barriers labeled A-D and segments R1-R3 represent example reaches between barrier “A” and other barriers in the stream network. A reach is defined as the link between two barriers denoted by i, the downstream barrier and j the upstream barrier. Barriers may be located between i and j, denoted by k, are binary removal decisions (B_k) in the stream network. My model uses a monthly timestep (m) for each objective. My study has 348 barriers, 66 on the mainstem Weber River and 282 in tributaries, with 121,104 potential

FIGURE 3. Conceptualized river network where boxes A-D represent barriers and R1-R3 represent example river reaches between barrier “A” and other barriers in the

..
reaches. Inputs into my model include a barrier penalty (0-1) determined by ability of a fish to move upstream or downstream from a barrier, quality-weighted connected habitat of each reach and water scarcity costs of large dams or diversions (Figure 4). My model is constrained by a removal budget. Development of the input data is described later in the text.

**FIGURE 4.** Conceptual diagram of optimization model maximizing quality-weighted habitat and minimizing water scarcity costs. Inputs to the model include economic water scarcity costs, costs of barrier removal, barrier passage and monthly quality-weighted connected habitat. Model outputs include sets of barrier removals.

### 3.6.1 Model Formulation

My weighted objective optimization method determined optimal barrier removal solutions between maximizing quality-weighted habitat and minimizing water scarcity to society for each month (m). A full list of the model notation is provided in Table A-1.
My first objective maximizes the Integral Index of Connectivity (IIC) with the quality-weighted habitat between barriers i and j (Equation 1). Here $A_i$ and $A_j$ are the longitudinal distance of quality weighted habitat above barriers i and j, respectively, $L_{i,j}$ is the topological distance between the two barriers and $CR_{ij}$ is the binary decision of reconnecting habitat between i and j by removing barrier, $B_k$. The second objective minimizes water scarcity costs ($c_k$) resulting from lost water deliveries (Equation 2). I maximized the combined objective function with a weight ($w$), summing to 1, applied on each objective to construct the Pareto-optimal frontier (Equation 3). Decisions in the model are barriers ($B_k$) to remove from the stream network and the passability ($P_j$ and $P_i$) representing a fish’s ability to pass beyond a barrier (0-1), where values of 1 are impassable barriers and 0.1 are completely passable. Impassable barriers were rated as 0.1 rather than 0 to avoid excluding passable barriers from barrier removal decision making.

Objective Functions:

\[
\text{Max } Z_{1m} = \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{A_i A_j}{1+L_{i,j}} \cdot CR_{ij} \cdot P_j \cdot P_i + \sum_{i=1}^{n} A_i^2
\]

\[
\text{Min } Z_{2m} = \sum_k c_k
\]

\[
\text{Max } Z_m = (1-w) \cdot Z1 - (w \cdot Z2)
\]

The $CR_{ij}$ term is defined as the sum of the barriers ($k$) between i and j plus the upstream barrier, j ($\text{Int}_{i,j,k}$) multiplied by the binary decision to remove barrier ($B_k$). This is divided by the sum of the barriers between i and j, plus the upstream barrier, j (Equation 4). For example, in Figure 3 barrier F is located between barrier E and G. If
barrier F is removed \( (B_{k=F}) \) then reach habitat above E and F are reconnected \( (CR_{i=E, j=F, k=F}) \). The \( CR_{ij} \) term is necessary to limit the count of overall connectivity to reaches free of all barriers.

The model also includes constraints representing physical, habitat, or economic bounds. Stream reach habitat suitability \( (A_k) \) is the spatially-intersected environmental variables discharge \( (Q_k) \), gradient \( (G_k) \), water temperature \( (T_k) \) and geomorphic condition \( (GC_k) \) (Table 6, Equation 5). Equation 6 specifies barrier removals are binary, thus a barrier is either fully removed or not removed. The total cost of barrier removal \( (TC) \) limits number of barriers removed based on cost of removing \( (C_k) \) barrier, \( B_k \) (Equation 7) and there is a binary decision to count a reach between barriers i and j (Equation 8).

Constraints:

\[
CR_{ij} \leq \sum_k \text{Int}_{i,j,k} * B_k / \sum_k \text{Int}_{i,j,k}, \ i \neq j \quad \text{Equation 4}
\]

\[
A_k = l_{Q_k \cap G_k \cap T_k \cap GC_k}, \ \forall k \quad \text{Equation 5}
\]

\[
B_k \{0,1\}, \ \forall k \quad \text{Equation 6}
\]

\[
TC \geq \sum C_k * B_k, \ \forall k \quad \text{Equation 7}
\]

\[
CR_{i,j} \{0,1\} \ \forall_{i,j} \quad \text{Equation 8}
\]

### 3.2 Habitat Suitability

#### 3.2.1 Reaches

I created the Weber Basin stream network in ESRI ArcGIS software with the National Hydrograph Dataset Plus Version 2 (NHD). I combined known in-stream barriers from NHD, USGS, and Trout Unlimited datasets to develop a barrier database. I segmented the stream network into reaches defined as stream length between barriers (Figure 3).
3.2.2 Discharge

I extracted average monthly NHD 1971-2000, gage adjusted streamflow to each reach in ArcGIS (U.S. Geological Survey, 2013). Figure 5 shows average 2005-2015 monthly flows from 13 stream gages in the Weber watershed, compared to estimated NHD flow. The NHD estimated flow compared to measured flow has a Standard Error of the Estimate (SEE) of 2.3 m$^3$/s (82.0 cfs), Percent Bias (PBIAS) of 29.5\%, $R^2$ of 0.96 and Root Mean Square Error of 2.3 m$^3$/s (81.5 cfs) (Table 10, Figure 5). At low flows NHD estimated discharge nears the one-to-one line, while at high flows the NHD estimates underestimate streamflow (Figure 5).

The Tennant method of environmental flows establishes flow conditions of river reaches by percent of mean annual discharge (Orth & Maughan 1981). The Tennant
method is the most widely used in-stream flow classification method (Gopal 2013; Pyrce 2004) and assumes a proportion of the mean annual discharge (MAD) is necessary to maintain healthy ecosystems. Observations of width, velocity and depth in 11 streams in Nebraska, Wyoming and Montana led to development of Tennant’s environmental flow method. Tennant’s flow recommendations stem from physical river characteristics and different flow quantity relationships to optimal fish habitat. 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 30% of MAD, while outstanding or optimum classification requires flows that are 60-100% of MAD (Table 2) (Gopal 2013; Jowett 1997; Orth & Maughan 1981). Table 2 displays the Tennant method of estimating in-stream flows by season. 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. Numerous variations of the Tennant method have been developed to apply the flow recommendations in different regions including British Columbia, Texas and Oklahoma (Gopal 2013; Linnansaari et al. 2012).

I created a modified version of the Tennant method for the Weber River Basin with classifications of poor, fair, good and excellent calculated from the percent of MAD (Table 3). I computed the MAD for each Strahler stream order and classification was computed with average 10 - 30 year historical flow data (prior to large dam and diversion development above the gage) (Table 4). I calculated the seasonal flow regime classification based on Strahler stream order.
TABLE 2. Tennant method to determine environmental flow conditions (Gopal 2013).

<table>
<thead>
<tr>
<th>In-stream Flow Classification</th>
<th>Recommended Flow Regimes (percent of Mean Annual Discharge)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>October-March (%)</td>
</tr>
<tr>
<td>Flushing or Maximum</td>
<td>200%</td>
</tr>
<tr>
<td>Optimum Range</td>
<td>60-100%</td>
</tr>
<tr>
<td>Outstanding</td>
<td>40%</td>
</tr>
<tr>
<td>Excellent</td>
<td>30%</td>
</tr>
<tr>
<td>Good</td>
<td>20%</td>
</tr>
<tr>
<td>Fair or Degrading</td>
<td>10%</td>
</tr>
<tr>
<td>Poor or Minimum</td>
<td>10%</td>
</tr>
<tr>
<td>Severe Degradation</td>
<td>0-10%</td>
</tr>
</tbody>
</table>


<table>
<thead>
<tr>
<th>Flow Classification</th>
<th>Recommended Flow Regimes (percent of Mean Annual Discharge)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>October-March (%)</td>
</tr>
<tr>
<td>Excellent</td>
<td>&gt;25%</td>
</tr>
<tr>
<td>Good</td>
<td>12-25%</td>
</tr>
<tr>
<td>Fair</td>
<td>5-12%</td>
</tr>
<tr>
<td>Poor</td>
<td>&lt;5%</td>
</tr>
</tbody>
</table>

TABLE 4. Weber Basin average maximum, minimum and average historical flows by Strahler stream order

<table>
<thead>
<tr>
<th>Strahler Stream Order</th>
<th>Historic Maximum Flow (m³/s)</th>
<th>Historic Minimum Flow (m³/s)</th>
<th>Historical Average Flow (m³/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.26</td>
<td>0.07</td>
<td>0.19</td>
</tr>
<tr>
<td>2</td>
<td>0.26</td>
<td>0.07</td>
<td>0.19</td>
</tr>
<tr>
<td>3</td>
<td>1.79</td>
<td>0.86</td>
<td>1.35</td>
</tr>
<tr>
<td>4</td>
<td>7.59</td>
<td>1.69</td>
<td>4.38</td>
</tr>
<tr>
<td>5</td>
<td>226.64</td>
<td>1.72</td>
<td>13.89</td>
</tr>
</tbody>
</table>
3.2.3 Water Temperature

Monthly water temperature was calculated from average monthly 2005-2015 PRISM 4km air temperatures and August 10-year average NorWeST stream temperatures (Isaak, D.J. et al. 2016, Prism Climate Group, 2016). Scully (2010) calculated that mean absolute error (MAE) of gridded PRISM estimated 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 scales because they are not spatially auto-correlated compared to daily time series (Caissie 2006; Erickson & Stefan 2000; Crisp & Howson 1982; Stefan & Preud’homme 1993). At high and low air temperatures, 0° C > TA > 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 & Stefan 1999). To account for patterns of spatial autocorrelation during relatively high August air temperatures, I obtained modeled August stream temperatures 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. 2016) (Table 9). For all other months (January-July and September-December), I linearly regressed stream temperatures from air temperatures (Equation 9).

\[ T_{k,t} = 4.2168 + 0.6259 \times (T_A, k, t) \]  \hspace{1cm} \text{Equation 9}

where \( T_{k,t} \) represents estimated average stream temperature (°C) during month, t and \( T_A, k, t \) is PRISM 10-year average air temperature (°C) between barriers i and j during month, t. I validated predicted stream temperature with observed 2015 average monthly
stream temperature. The 2015 observed versus predicted water temperatures have an 
$R^2$ of 0.93, MAE of 1.28 °C, RMSE of 1.55 °C, and percent bias (PBIAS) of 2% (Figure 
6).

I categorized stream temperatures for Bonneville cutthroat trout as poor, fair, 
good or excellent. Poor water temperatures are over 21°C and excellent water 
temperatures are under 15 °C (Table 6) (Schrank et al. 2003; Cade 1985).

3.2.4 Gradient

I estimated gradient with a digital elevation model (DEM) in GIS. Excellent 
habitat is considered between 0-6% gradients while poor habitat is over 10% gradients
(T) (Kershner 1992; Rosenfeld et al. 2000; Hilderbrand & Kershner 2004).

FIGURE 6. Predicted versus observed average monthly water temperature. Red dotted 
line is a one to one relationship.
3.2.5 Geomorphic Condition

Stream reach geomorphic conditions, developed by the Fluvial Habitat Center at Utah State University for the Weber River, range from intact or undisturbed, to poor or severely impaired and degraded (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 5, Portugal et al. 2016).

<table>
<thead>
<tr>
<th>Table of Geomorphic Conditions</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Intact</strong></td>
</tr>
<tr>
<td><strong>Good</strong></td>
</tr>
<tr>
<td><strong>Moderate</strong></td>
</tr>
<tr>
<td><strong>Poor</strong></td>
</tr>
</tbody>
</table>

3.2.6 Habitat Suitability Classification

I intersected percent of mean annual discharge, monthly water temperature, gradient, and geomorphic condition of each stream reach in a GIS database and classified reaches into excellent, good, fair, and poor habitat suitability categories (Table 6, Figure 8). August habitat suitability is demonstrated in Figure 8 all other months are in Appendix B. A reach with excellent Bonneville cutthroat trout habitat with a rating of 1, met all conditions of: gradient <6%, good or intact geomorphic condition, water temperature <15°C and discharge >25% the mean annual discharge between October and
March, and >60% of mean annual discharge between April through September. A reach with poor habitat received a rating of 0.1 if any of the following occurred: water temperature >21°C, gradient >10%, and discharge less than 5% of the mean annual discharge. I assigned a poor habitat rating of 0.1 rather than 0, to ensure the barrier value remains above zero when multiplying the passage penalty within the equation. A barrier value of zero would remove the barrier as a removal option from decision-making.

Lindley et al. (2006) and Null et al. (2014) previously used a similar habitat suitability classification for steelhead trout in California streams. Additionally, numerous studies applied quality habitat classification and scoring for fish species in other watersheds (Nunn & Cowx 2012; Burnett et al. 2003; Quist et al. 2005) (Table 6).

I compared classifications of stream reaches with suitable habitat to known populations of Weber Basin Bonneville cutthroat trout (Figure 7) using the Fisher’s exact test. Known population estimates from Trout Unlimited provide a general idea of fish population but are preliminary data and do not vary seasonally. The p-value of < 0.001, suggests that the habitat suitability ratings are significant in predicting observed fish counts. The habitat suitability accuracy in Figure 7 identify where the classification is accurate, overestimating or underestimating August habitat classification. Habitat suitability is an overestimate when a reach is classified as good or excellent but does not contain a large fish population. A reach is an underestimate if the classification is poor or fair but contains a large fish population.
FIGURE 7. Observed Bonneville cutthroat trout count data compiled by Trout Unlimited and Utah Division of Wildlife Resources compared to estimated August habitat suitability. Habitat suitability is an overestimate when a reach is classified as good or excellent but does not contain a large fish population. A reach is an underestimate if the classification is poor or fair but contains a fish population.

I calculated each reach’s longitudinal length (km) in ArcGIS. I multiplied reach length (HL_{k,t}) by the habitat rating (Hql_{k,t}) (Equation 2) for each stream reach, to determine quality-weighted habitat (Hw_{k,t}) between barriers i and j for each month, t (Table 6, Figure 8).

\[ Hw_{k,t} = Hql_{k,t} \times Hl_{k,t} \]  
Equation 10
TABLE 6. Habitat categories and criteria to determine Bonneville cutthroat trout habitat suitability. All criteria must be met for excellent, good and fair habitat categories and any of the criteria must be met for poor habitat.

<table>
<thead>
<tr>
<th></th>
<th>Water Temperature (°C)</th>
<th>Gradient (%)</th>
<th>Flow October-March (% of MAD)</th>
<th>Flow April-September (% of MAD)</th>
<th>Geomorphic Conditions</th>
<th>Rating</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Excellent</strong></td>
<td>&lt; 15</td>
<td>0 - 6</td>
<td>&gt; 25%</td>
<td>&gt; 60%</td>
<td>Good or Intact</td>
<td>1</td>
</tr>
<tr>
<td><strong>Good</strong></td>
<td>&lt; 18</td>
<td>0 - 9</td>
<td>12% - 25%</td>
<td>40% - 60%</td>
<td>Good or moderate or intact</td>
<td>0.75</td>
</tr>
<tr>
<td><strong>Fair</strong></td>
<td>&lt; 21</td>
<td>0 - 10</td>
<td>5% - 10%</td>
<td>10% - 40%</td>
<td>good or poor or moderate or intact</td>
<td>0.25</td>
</tr>
<tr>
<td><strong>Poor</strong></td>
<td>&gt; 21</td>
<td>&gt; 10</td>
<td>&lt; 5%</td>
<td>&lt; 10%</td>
<td>good or poor or moderate or intact</td>
<td>0.10</td>
</tr>
</tbody>
</table>
Habitat criteria limited a reach if it prevented the classification from moving into a higher category. For example, a reach with August water temperature of 20 °C, gradient of 6%, August discharge of 50% MAD, and good geomorphic conditions classified as “Fair” habitat. The limiting habitat criteria were water temperature because it prevented the August habitat classification from moving into the “Good” class. August limiting criteria is shown in Figure 9, all other months are in Appendix C.
3.3 Habitat Connectivity

Numerous metrics of connectivity have been suggested to quantify longitudinal river habitat connectivity (Pascual-Hortal & Saura 2006; Freeman 1977; Jaeger 2000; Cote et al. 2009; Erős et al. 2012; Grill et al. 2014). Many habitat patch connectivity indices use graph theory, relating barriers and links to represent stream reaches and model longitudinal connectivity between habitat patches (Schick & Lindley 2007; Erős et al. 2012; Eros et al. 2011; Saura & Rubio 2010; Urban & Keitt 2001). Among the proliferation of metrics available, one of the most suitable for predicting impact of fragmentation in river networks include the Integral Index of Connectivity (IIC) (Malvadkar et al. 2015). The IIC measures the degree of habitat connectivity at the watershed-scale, ranging from 0- no connection to 1- fully connected watershed absent of...
barriers (Pascual-Hortal & Saura 2006). The IIC takes into account topological
distance \((L_{ij})\) of habitat patches between barriers \(i\), and \(j\). Variables \(A_i\) and \(A_j\) represent
quality-weighted habitat length of stream reaches \(i\) and \(j\) and \(A_L\) is total stream habitat
length (Malvadkar et al. 2015; Pascual-Hortal & Saura 2006). The IIC, maximizing
reconnected habitat is directly optimized in the objective function but the count of overall
connectivity is limited to unimpeded stream lengths. Equation 11 gives the habitat
connectivity value:

\[
Z_{1m} = \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{A_i A_j}{L_{ij} + 1} \cdot CR_{ij} \cdot P_j \cdot P_i + \sum_{j=1}^{n} A_j^2, \quad i \neq j
\]  

Equation 11

Each barrier has a corresponding upstream quality-weighted habitat distance. The
ideal reach length (unimpeded length of river between two barriers) was defined as the
entire watershed but different reach lengths were modeled based on previous studies of
Bonneville cutthroat trout habitat ranges (Schrank & Rahel 2004; Carlson & Rahel 2010;
Colyer et al. 2005; Young 2011). For example, sets of barrier removals increasing the
overall habitat were prioritized. If the ideal reach length was 30 km, habitat above
barriers farther than 30 km were considered disconnected habitat (Saura, S.& J. Torné.
2009).

3.4 Barrier Passage

I assigned each barrier a passage rating based on the probability of Bonneville
cutthroat trout moving beyond the barrier throughout an entire year. Fish passage weights
are from previous ratings from a Trout Unlimited study where potential and known
barriers were categorized and given a passage classification with expert knowledge and areal imagery. I further refined the passage rating of the identified barriers from stream gradient, stream order, culvert length and areal imagery (Table 7). I based rating scores on previous classification systems where zero was not passable, 0.3 was partially not passable, 0.6 was mostly passable, and 1 was completely passable (King & O’Hanley 2016; Scotland & Northern Ireland Forum for Environmental Research, Edinburgh 2010).

A barrier is partially passable if a fish can move past the barrier given the appropriate hydrologic conditions for their life stage. If a culvert was located on a stream order 4 or less, I considered it partially passable (Neeson et al. 2015), otherwise, I assessed the culvert length and gradient. A culvert on a stream reach over 6% was not passable, 5-6% was partially un-passable, 4-5% was partially passable and 4% was passable (Poplar-Jeffers et al. 2009). I rated a culvert greater than 85 m as not passable and under 11 m, completely passable (Weaver 1963; Warren & Pardew 1998; King & O’Hanley 2016). Box culverts greater than 750 m I rated not passable and under 100 m as completely passable. Lastly, turbulence beneath barriers, identified in areal imagery, estimated hydraulic drop for unknown barriers (Scotland & Northern Ireland Forum for Environmental Research, Edinburgh 2010). I compared passage ratings to those found in the Trout Unlimited analysis. Ratings that differed I validated depending on known information from expert opinion regarding the barrier passability. For example, if a barrier is known to be impassable for fish movement upstream and downstream but rated otherwise, I changed the passage rating.
TABLE 7. Criteria for barrier passage classifications. I rated barrier passage by culvert length, water turbulence, stream order, gradient and expert opinion (Trout Unlimited Study).

<table>
<thead>
<tr>
<th></th>
<th>Slope (reach) - GIS derived</th>
<th>Strahler order - GIS derived</th>
<th>Length of Culvert (m)</th>
<th>Box Culvert Length (m)</th>
<th>Water turbulence for all structures</th>
</tr>
</thead>
<tbody>
<tr>
<td>1, passable</td>
<td>&lt; .04</td>
<td>&gt; 5</td>
<td>&lt;= 10</td>
<td>&lt;= 100</td>
<td>low</td>
</tr>
<tr>
<td>0.6, mostly passable</td>
<td>.04 - .05</td>
<td>&lt;= 4</td>
<td>11 - 30</td>
<td>100 - 400</td>
<td>moderate</td>
</tr>
<tr>
<td>0.3, partially not passable</td>
<td>.05 - .06</td>
<td>&lt;= 4</td>
<td>31 - 85</td>
<td>400 - 750</td>
<td>high</td>
</tr>
<tr>
<td>0, not passable</td>
<td>&gt; .06</td>
<td>&lt;= 4</td>
<td>&gt; 85</td>
<td>&gt;= 750</td>
<td>high</td>
</tr>
</tbody>
</table>

I incorporated barrier passage ratings into the model as a barrier penalty. I flipped passage probability scores to assign barrier penalties so that passable barriers scored as 1 were given a passage penalty of 0.10 and impassable barriers classified as 0 were given a passage penalty of 1. I assigned higher penalties to un-passable barriers and lower penalties to less obstructive barriers to nudge the model to remove more inhibitive barriers.

3.5 Removal Costs

I estimated culverts, diversions and dam removal costs from American River’s database, expert opinion and previous barrier removal and restoration projects. Table 8 shows the maximum, minimum and average barrier removal costs. I estimated culvert
removal costs from known culvert length or measured culvert length in areal imagery. Culverts between 20 and 50 ft long (6.1 – 15.2 m), presumed a two lane road, I estimated at $150,000 while over 50 ft (15.2 m), assumed at least a four lane road, I estimated at $75,000 per lane or $300,000 (Neeson et al. 2015; Salt Lake City Department of Public Utilities 2008). Removal costs of culverts under 20 ft (6.1 m), I calculated with an equation developed by Dupont (2000) based off experience with culvert removals in Idaho. Dupont (2000)’s equation relates culvert length (CL) and cost of building materials, adjusted for inflation, to estimate removal costs (Equation 12).

\[
\text{Cost} = 33500 + 804 \times \text{CL} \quad \text{Equation 12}
\]

I estimated diversion removal costs from expert opinion and if known, diverted water quantity and diversion structure size. Large diversion removal costs, primarily used for municipal water use with capacity of 28.3 m$^3$/s (1000 cfs) or more I estimated at $1 M (per comm. Paul Bernett Trout Unlimited 2016). Small diversion removal costs, less than 28.3 m$^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 Database, 2015). Dams with an unknown height I assigned the average cost ($250,000) of barriers removed between 1 and 5 ft (0.3 – 1.5m) high. I compared Klamath Dam removal project cost estimates to Weber Basin large dams height, length and reservoir capacity (US Dept. of Interior et al. 2012). I estimated removing large dams in the Weber Basin would cost $30 M, except the largest reservoir in the basin, Pineview Dam, which I estimated at $50 M.
3.6 Water Scarcity Costs

As population in the Wasatch Front and Weber watershed continue to grow, it is increasingly important to consider economic water uses in water resources modeling. Economic water uses, including water supply, hydropower, flood protection and irrigation are monetized depending on water supply and demand. Managing water resources as an economic good enables efficient resource management to mitigate water scarcity (Van der Zaag et al. 2006). Valuing water use considers water demands and prices of water delivered by quantity, water use, and time of year (Jenkins et al. 2003; Harou et al. 2009). Economic loss functions represent dynamic costs of water, estimating prices that residential, commercial, industrial and institutional water users would be willing to pay for additional water (Figure 10) (Jenkins et al. 2003; Whitelaw & Macmullan 2014; Harou et al. 2009; Van der Zaag & Savenije 2006; Draper et al. 2003). Economic loss functions aggregate water demand and apply a price elasticity, which is the change in quantity per change in price of water (Harou et al. 2009) to construct a water demand curve. Null et al. (in prep) developed Ogden and Wasatch Front seasonal economic water use demand curves, presented in Figure 10. Under the water demand function, water deliveries that meet or exceed target water demands result in no water scarcity losses (also referred to as economic losses). When water deliveries are less than the demand, water scarcity represents costs incurred to users (Jenkins et al. 2003). During

<table>
<thead>
<tr>
<th></th>
<th>Minimum ($)</th>
<th>Maximum ($)</th>
<th>Average ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diversions</td>
<td>300,000</td>
<td>50,000,000</td>
<td>569,792</td>
</tr>
<tr>
<td>Impoundments</td>
<td>250,000</td>
<td>30,000,000</td>
<td>2,662,162</td>
</tr>
<tr>
<td>Road Crossings</td>
<td>36,555</td>
<td>300,000</td>
<td>131,244</td>
</tr>
</tbody>
</table>
summer months, water demands are greater, resulting in increased losses compared to the same amount of water delivered during other times of the year.

I estimated agriculture and urban economic loss with water demand and delivery cost functions for seven water supply reservoirs and three major diversions (Null et al. in prep). To estimate water scarcity losses, I defined 30-year average monthly flow downstream of reservoirs equal to water demands, resulting in zero water scarcity costs. 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, while 5% water deliveries ranged between $129 M and $856 M for the watershed, depending on the season. Large diversion removal resulted in 100% lost water deliveries. I assumed without the diversion no water could be delivered through the canal or pipeline, thus resulting in 0% of water deliveries.

FIGURE 10. Seasonal demand function for the Ogden urbanized area (Null, unpublished). The water demand is split by seasons, Summer (May-September), Winter (November-March) and Intermediate months (October, April).
3.7 Model Runs

I implemented the model at a monthly time-step for each objective weight and budget level. I modeled different alternatives (Table 9) and removal budgets to identify promising barriers to remove, and graphically and spatially interpret tradeoffs between economic losses, removal costs and quality-weighted connected habitat.

<table>
<thead>
<tr>
<th>30km Connected Reach Length</th>
<th>Dispersal Threshold of the Entire Watershed</th>
<th>Maximizing Quality-Weighted Habitat (Connectivity not included)</th>
<th>Without Barrier Passage</th>
<th>Not including economically important barriers</th>
<th>50% Increase to Removal Costs</th>
<th>50% Decrease to Removal Costs</th>
<th>Single Objective: Maximizing the Quality-Weighted Connected Habitat</th>
<th>One Removal Limit</th>
<th>Five Removal Limit</th>
<th>One Removal Limit- no economic loss</th>
<th>Five Removal Limit- no economic loss</th>
</tr>
</thead>
</table>

TABLE 9. Optimization model scenarios
<table>
<thead>
<tr>
<th>Variable</th>
<th>Model</th>
<th>Description</th>
<th>Units</th>
<th>Range</th>
<th>Error</th>
<th>Citations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Air Temperature</td>
<td>Stream Temperature</td>
<td>PRISM gridded 10 year average monthly air</td>
<td>°C</td>
<td>Monthly</td>
<td>4km gridded resolution</td>
<td>Prism Climate Group, 2016</td>
</tr>
<tr>
<td></td>
<td></td>
<td>temperature</td>
<td></td>
<td></td>
<td>MAE = 0.72°C to 0.74°C Mean Bias Error=</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>-0.11°C to 0.13°C</td>
<td></td>
</tr>
<tr>
<td>NorWeST Stream Temperature</td>
<td>Stream Temperature</td>
<td>NorWeST calculated August water temperature</td>
<td>°C</td>
<td>August</td>
<td>RMSPE=1.07°C MAE= 0.74°C</td>
<td>Isaak, D.J. et al. 2016.</td>
</tr>
<tr>
<td></td>
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<td></td>
<td></td>
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</tr>
<tr>
<td>Stream Temperature</td>
<td>Habitat Connectivity</td>
<td>Calculated water temperature</td>
<td>°C</td>
<td>Monthly</td>
<td>RMSE= 1.55 °C MAE = 1.28 °C PBIAS= 2%</td>
<td>Caissie 2006; Erickson &amp; Stefan 2000; Crisp &amp; Howson 1982; Stefan &amp; Preud’homme 1993; Mohensi &amp; Stefan 1999</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>R²= 0.93</td>
<td></td>
</tr>
<tr>
<td>Flow</td>
<td>Habitat Connectivity</td>
<td>Gage adjusted stream flow from 1971-2000</td>
<td>m³/s</td>
<td>Monthly</td>
<td>SEE= 2.6 m³/s PBIAS=29.5% R²=0.96 RMSE= 81.45</td>
<td>U.S. Geological Survey, 2013</td>
</tr>
<tr>
<td>TABLE 10. (cont.)</td>
<td></td>
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<td>-------------------</td>
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<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Gradient</td>
<td>Habitat Connectivity</td>
<td>Reach gradient</td>
<td>%</td>
<td>0-22</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Geomorphology</td>
<td>Habitat Connectivity</td>
<td>River geomorphic condition</td>
<td>unitless</td>
<td>Intact, Good, Fair, Poor</td>
<td>NA</td>
<td>Portugal, E.W., et al. 2016</td>
</tr>
<tr>
<td></td>
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<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Cost of Barrier Removal</td>
<td>Optimization</td>
<td>Cost of removing a culvert, diversion or impoundment</td>
<td>$</td>
<td>0-50 $M</td>
<td>NA</td>
<td>Neeson et al. 2015; Salt Lake City Department of Public Utilities 2008; per comm. Paul Bernett Trout Unlimited; Klamath Dam Report; per comm. Mitigation Commission Provo Diversion; American Rivers Database</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Economic Costs</td>
<td>Optimization</td>
<td>Economic loss associated with removing a large dam</td>
<td>$</td>
<td>0-856 $M</td>
<td>NA</td>
<td>Jenkins et al. 2003; Whitelaw &amp; Macmullan 2014; Harou et al. 2009; Null et al. unpublished</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Discharge with and without dams</td>
<td>Economic</td>
<td>Average discharge above and below large dams</td>
<td>m³/s</td>
<td>monthly</td>
<td>NA</td>
<td><a href="https://waterdata.usgs.gov/ut/nwis/current/?type=flow">https://waterdata.usgs.gov/ut/nwis/current/?type=flow</a></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habitat Connectivity</td>
<td>Optimization</td>
<td>Barrier contribution to habitat connectivity</td>
<td>%</td>
<td>0-1</td>
<td>NA</td>
<td>Pascual-Hortal and Saura 2006; Mavadkar 2015; Branco 2014; Eros et al. 2011; Saura, S. &amp; J. Torné. 2009</td>
</tr>
</tbody>
</table>
CHAPTER IV

RESULTS

4.1 Habitat Benefits of Barrier Removal versus Removal and Water Scarcity Costs

After reconnecting about 500 km of habitat, removal of barriers resulting in water scarcity costs begins. Initially, water scarcity costs are about $29,000 per 1 km reconnected habitat. Near a budget of $40 M, water scarcity costs increase to about $1 M per 1 km reconnected habitat. Water scarcity costs from barrier removal do not begin until higher budget levels because removal costs are greater for large economically important barriers compared to small barriers.

More than 500 km of quality-weighted, connected habitat can be added in August by removing small in-stream barriers without affecting water supply or incurring water scarcity costs (Table 11, Figure 11). This entails removing 337 barriers, with total barrier removal costs of just over $83 million (Figure 12). The model prioritizes removing economically costly barriers after nearly all other barriers are removed. Water scarcity costs exceed $660 million when an additional 10 water supply dams and diversions are removed, adding an additional 124 km of habitat. Table 11 displays results from a selected optimization model run for August. The model was ran for 12 independent habitat suitability scenarios representing habitat in each month (Table 12), but the analysis focused on August habitat conditions because August constrains Bonneville cutthroat trout populations (Carlson & Rahel 2010; Colyer et al. 2005; Young 2011). In Figure 11, there is the most reconnected habitat (objective weight of 0) when water scarcity costs are the greatest.
For the first 160 km (26%) and $10 M in barrier removal costs, habitat gain increases substantially, after the least expensive barriers are removed, the cost effectiveness of removing barriers to connect river habitat slows (Figure 12). At low levels of reconnected habitat of Figure 12, a budget of $89,600 reconnects 8 km of habitat or $11,200 per 1 km reconnected habitat, but at higher budget levels, a 10 km habitat gain costs an additional $20 million or $363,700 per 1 km habitat. In other words, there is decreasing marginal benefit of removing barriers, so that after the first 54 barriers are removed, it costs more to gain the same length of habitat. When water scarcity costs and budget removal costs are compared (Figure 13), water scarcity losses begin at a budget of $10 M when maximizing habitat is the primary objective. As the minimize water scarcity costs objective is given a larger weight, water scarcity costs are incurred at higher budget levels. At a 75% weight maximizing habitat connectivity (25% minimizing

FIGURE 11. Tradeoff curve for August habitat gain versus water scarcity costs. Curve is for August habitat gain.
water scarcity), barriers resulting in water scarcity are not removed until a budget of $80 M. At equal objective weights economically important barriers are never removed, despite a sufficient budget. The tradeoff between the index of connectivity and budget is nearly linear. Initially, the basin connectivity index is 0.065 and increases by about 0.06 per $10M. At higher budget levels the rate of connectivity decreases to 0.012 per $10M (Figure 14).

**TABLE 11.** Water scarcity costs, habitat gain and number of barriers removed by objective weight for August habitat suitability.

<table>
<thead>
<tr>
<th>Weight on Economic Objective</th>
<th>Economic Loss ($)</th>
<th>Habitat Gain (km)</th>
<th>Barriers Removed</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.02</td>
<td>607932442</td>
<td>602</td>
<td>347</td>
</tr>
<tr>
<td>0.25</td>
<td>31678782</td>
<td>515</td>
<td>339</td>
</tr>
<tr>
<td>0.50</td>
<td>0</td>
<td>478</td>
<td>337</td>
</tr>
<tr>
<td>0.75</td>
<td>0</td>
<td>478</td>
<td>337</td>
</tr>
<tr>
<td>0.98</td>
<td>0</td>
<td>478</td>
<td>337</td>
</tr>
</tbody>
</table>

**TABLE 12.** Barrier removal costs, water scarcity costs, habitat gain and number of barriers removed by month.

<table>
<thead>
<tr>
<th>Month</th>
<th>Barrier Removal Costs ($)</th>
<th>Water Scarcity Costs ($)</th>
<th>Reconnected Habitat (km)</th>
<th>Barriers Removed</th>
</tr>
</thead>
<tbody>
<tr>
<td>January</td>
<td>99,992,255</td>
<td>0</td>
<td>402</td>
<td>162</td>
</tr>
<tr>
<td>February</td>
<td>99,997,981</td>
<td>0</td>
<td>385</td>
<td>161</td>
</tr>
<tr>
<td>March</td>
<td>99,985,346</td>
<td>0</td>
<td>393</td>
<td>163</td>
</tr>
<tr>
<td>April</td>
<td>99,973,869</td>
<td>0</td>
<td>416</td>
<td>168</td>
</tr>
<tr>
<td>May</td>
<td>99,973,869</td>
<td>0</td>
<td>423</td>
<td>168</td>
</tr>
<tr>
<td>June</td>
<td>99,964,123</td>
<td>3,397,428</td>
<td>422</td>
<td>169</td>
</tr>
<tr>
<td>July</td>
<td>99,996,764</td>
<td>10,068,912</td>
<td>537</td>
<td>283</td>
</tr>
<tr>
<td>August</td>
<td>83,862,022</td>
<td>0</td>
<td>478</td>
<td>337</td>
</tr>
<tr>
<td>September</td>
<td>100,000,000</td>
<td>10,068,912</td>
<td>464</td>
<td>284</td>
</tr>
<tr>
<td>October</td>
<td>99,964,123</td>
<td>0</td>
<td>405</td>
<td>169</td>
</tr>
<tr>
<td>November</td>
<td>99,973,869</td>
<td>0</td>
<td>381</td>
<td>168</td>
</tr>
<tr>
<td>December</td>
<td>99,982,267</td>
<td>0</td>
<td>395</td>
<td>167</td>
</tr>
</tbody>
</table>
FIGURE 12. Tradeoff curve for August habitat gain versus removal budget with equal weights on both objectives. Initially, reconnected habitat costs $11,200 per 1 km, but at higher budgets increases to $363,700 per 1 km of reconnected habitat.

FIGURE 13. Comparison between water scarcity and barrier removal costs. When the quality-weighted habitat is given priority, barriers resulting in economic loss are removed with a budget of $10 M. As the minimizing water scarcity objective gains emphasis, economically important barriers are not removed until higher budget levels. At equal objective weights, no barriers resulting in economic loss are removed.
Figure 14. Tradeoff between the index of connectivity and budget at equal objective weights

Plotting the longest connected stream length compared to the budget displays a similar trend as the budget and upstream habitat tradeoff. Initially, the longest single reach is 36 km with $184,200 per 1 km reach length added to the longest reach and increases to a maximum of 142 km or $3,030,300 per 1 km with equal objective weights (Figure 15). Given an $80 M budget, the maximum reach length is 144 km greater (286 km) when prioritizing the quality-weighted objective compared to equal weights and when minimizing water scarcity costs (Figure 16). The average reach length is shortest when maximizing quality-weighted connected habitat (4 km) and highest when prioritizing minimizing water scarcity costs (24 km).
FIGURE 15. Tradeoff between maximum reach length and budget. The maximum reach length is defined at the longest connected reach given a budget.

FIGURE 16. Reach length with different objective weights using a $80 M budget. The red dot represents the average reach length and the maximum reach lengths are labeled.
4.2 Sensitivity Analyses

Input and model objectives were varied to demonstrate how stable results are to uncertainty of inputs and objectives. A summary of model runs and results using a $100 M budget and August habitat suitability are presented in Table 13. Not including barriers that result in water scarcity resulted in slightly lower habitat gains than scenarios with water scarcity losses. When optimizing a single objective, maximizing the quality-weighted connected habitat, economic losses exceeded $400 M compared to dual objective model runs because barriers integral for water supply were removed. Two scenarios included an additional constraint on number of barriers the model could remove.

4.2.1 Sensitivity Analysis: Effect of the Connectivity Index

Optimizing quality-weighted habitat without including an index of connectivity resulted in more water scarcity and quality-weighted habitat compared to model runs with the connectivity index. However, quality-weighted habitat was fragmented throughout the watershed, scoring 19% less in overall habitat connectivity (Table 13). The cost effectiveness of removing barriers declined considerably beyond reconnecting 382 km and removing 137 barriers (Figure 17). When comparing reconnected habitat with and without an index of connectivity, the Pareto efficiency tradeoff curve differs between removal budgets below approximately $100 M (Figure 18). Near the maximum difference between the curves at 333 km reconnected habitat, barrier removal without considering connectivity cost $21 M less than when habitat connectivity is included. Increasing quality-weighted connected habitat came as a tradeoff with disconnected
habitats (Table 13), which could be valued for non-migratory species. With the connectivity index, model results centralized and aggregated

**TABLE 13. August model results with a $100 M barrier removal budget.**

<table>
<thead>
<tr>
<th>Modeling Scenario</th>
<th>Economic Loss ($)</th>
<th>Habitat Gain (km)</th>
<th>Spent ($)</th>
<th>Connectedness (%)</th>
<th>Number Of Barriers Removed</th>
<th>weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>80km Dispersal Radius</td>
<td>0</td>
<td>478.3</td>
<td>83,862,022</td>
<td>46%</td>
<td>337</td>
<td>0.5</td>
</tr>
<tr>
<td>Dispersal Threshold of the Entire Watershed (1182km)</td>
<td>0</td>
<td>478.3</td>
<td>83,862,022</td>
<td>46%</td>
<td>add</td>
<td>0.5</td>
</tr>
<tr>
<td>Maximizing Quality-Weighted Habitat without an Index of Connectivity</td>
<td>307,317,369</td>
<td>499.8</td>
<td>99,998,328</td>
<td>27%</td>
<td>169</td>
<td>0.5</td>
</tr>
<tr>
<td>Single Objective: Maximizing the Quality-Weighted Connected Habitat (Not Including Minimizing Water Scarcity Costs)</td>
<td>406,098,992</td>
<td>99,988,341</td>
<td>62%</td>
<td>269</td>
<td>NA</td>
<td></td>
</tr>
<tr>
<td>Not Including Barrier Passage</td>
<td>17,176,910</td>
<td>515.5</td>
<td>99,980,059</td>
<td>53%</td>
<td>276</td>
<td>0.5</td>
</tr>
<tr>
<td>Not including economically important barriers</td>
<td>0</td>
<td>478.3</td>
<td>83,862,022</td>
<td>46%</td>
<td>337</td>
<td>NA</td>
</tr>
<tr>
<td>50% Decrease to Removal Costs</td>
<td>0</td>
<td>478.3</td>
<td>83,862,022</td>
<td>46%</td>
<td>337</td>
<td>0.5</td>
</tr>
<tr>
<td>50% Increase to Removal Costs</td>
<td>0</td>
<td>407.6</td>
<td>49,998,445</td>
<td>33%</td>
<td>209</td>
<td>0.5</td>
</tr>
<tr>
<td>One Removal Limit</td>
<td>0</td>
<td>7.9</td>
<td>250,000</td>
<td>7%</td>
<td>1</td>
<td>0.5</td>
</tr>
<tr>
<td>Five Removal Limit</td>
<td>0</td>
<td>43.4</td>
<td>1,250,000</td>
<td>7%</td>
<td>5</td>
<td>0.5</td>
</tr>
<tr>
<td>One Removal Limit-no economic loss</td>
<td>NA</td>
<td>11.0</td>
<td>150,000</td>
<td>7%</td>
<td>1</td>
<td>0.5</td>
</tr>
<tr>
<td>Five Removal Limit-no economic loss</td>
<td>NA</td>
<td>35.7</td>
<td>1,250,000</td>
<td>7%</td>
<td>5</td>
<td>0.5</td>
</tr>
</tbody>
</table>
FIGURE 17. Tradeoff between budget and quality-weighted habitat. Near a budget of $30 M and 382 km of connected habitat, the rate of habitat gain per dollar reduces.

FIGURE 18. August tradeoff curve of barrier removal budget versus total habitat gain with and without a connectivity index for quality-weighted habitat.
The ideal reach length influences the location and type of barrier removal. Adjusting ideal reach lengths connects areas of different sizes so would affect dispersal or migration distances for different life stages or species. For example, an ideal connectivity reach length of 30 km encompasses a barrier’s connectivity within a 30 km distance. To see how the connectivity threshold changed barrier removal results, connectivity reach lengths were varied between 30 km and the total connected quality-weighted habitat length (577,145 km to 779,176 km depending on month). There was no significant difference between reach lengths with ideal reach length over 30km, but below 30 km the type and location of barriers removed varied between connectivity reach lengths. The top 15 barriers in August habitat suitability and equal objective weights are
diversions and road crossings located in tributary reaches (Figure 20, Figure 21).

Modeling with smaller connectivity thresholds suggests that removing road crossings and barriers located in tributaries is optimal. Smaller connected reaches are beneficial to species that do not migrate or that require small habitat patches.

Comparing barrier removal alternatives at $100 M budget, habitat gain did not differ substantially between connectivity reach lengths (Table 13); although, the portfolio of removed barriers differed between scenarios. At a budget of $1.5 M, the set of barriers removed with different connectivity reach lengths changed somewhat, but each run contained commonly removed barriers (Figure 22). Similar to the most common barriers removed, model results prioritized barriers in tributaries in all connectivity reach lengths.

FIGURE 20. Top 15 barrier types removed with varying connectivity thresholds. Count includes barriers removed for all months and weights.
FIGURE 21. Location of top 15 barriers removed with varying connectivity thresholds. Count includes number of barriers removed for all months and weights.

FIGURE 22. Barrier removal results at a $1.5 M budget in 30km ideal connectivity reach lengths and the entire watershed.
4.2.2 Types of barriers removed

The type of barriers removed (diversion, road crossing or impoundment) varied by barrier removal budget levels (Figure 23). These results differ from Figure 20 by including total count at each budget level. Removing road crossings and diversions occurred prior to large dams for all scenarios, likely because dams cost more to remove and could increase water scarcity costs. The model removed diversions more commonly with all budgets compared to other types of barriers, but this is unsurprising because diversions account for nearly 50% of the total barriers. With budgets of less than $100 M, the model removed most of the barriers, prioritizing the less expensive and economically neutral barriers over the more expensive and economically adverse impoundments (Figure 23).

FIGURE 23. Type of barriers removed with different budgets.
4.2.3 Dam Removal Costs

Increasing or decreasing dam removal costs adjusts the inflection point and slope of the tradeoff curve. An increase in costs increases the slope, while decreasing costs flattens the slope between habitat gain and budget (Figure 24). The inflection point where the marginal benefit of removing barriers decreases ranges between $5 M (50% cost reduction) to about $15 M (150% increase in barrier removal costs).

FIGURE 24. Habitat gains with different barrier removal budgets
Tradeoffs exist between quality-weighted aquatic habitat connectivity and water scarcity costs. Initially, the marginal cost of habitat is about $11,200 per 1 km habitat gain but as the least expensive barriers are removed, the marginal cost increased to $363,700 per 1 km habitat gain. Identifying ideal investment and economic threshold levels to gain the most habitat is important for making cost efficient barrier removal decisions. My barrier removal cost estimates per kilometer habitat gain are in the same ballpark as past research on small barrier removal and restoration (Bernhardt et al. 2005; Reagan 2015; O’Hanley & Tomberlin 2005). For example, Wait et al. (2004) found costs ranged from $13,120 – 305,920 per kilometer of opened habitat in Washington streams.

When both objectives are weighted equally, the model never removes large, economically-important barriers despite an adequate budget. Removing only small, non-economically important barriers reconnects 119 km of quality-weighted habitat by about by removing 54 barriers. If more weight is given to maximizing quality-weighted connected habitat, Stoddard Diversion is removed with a barrier removal budget of $80 M (Figure 25). If quality-weighted connected habitat is weighted significantly more (98%) than minimizing water scarcity, Stoddard Diversion is removed with a budget of $10 M (Figure 25). Stoddard Diversion is the only large barrier removed until the barrier removal budget reaches $70 M.

Removing numerous small barriers does not affect water supply or incur water scarcity costs and connects quality-weighted habitat at the least cost, compared to
including economically-expensive barriers (Table 13). Thus, focusing on small barrier removal is potentially effective to improve habitat connectivity while minimizing water scarcity costs. Model results indicate that it is ideal to remove water supply barriers if society gives greater priority to aquatic habitat over water supply. 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-

FIGURE 25. Budgets and reconnected habitat tradeoffs when large barriers are removed. Tradeoff curve (bottom) and barriers removed (top) are for August habitat suitability and 85% weight on quality-weighted connected habitat.
making. It is never optimal to remove water supply barriers if water supply is prioritized over aquatic habitat. While the latter has historically been prioritized (especially in arid, semi-arid, Mediterranean climates), large-scale reductions in habitat, species, ecosystem services, and water quality have led to recent notable instances where habitats were prioritized over water supply (Kruse et al. 2006; US Dept. of Interior et al. 2012).

It is beneficial to include economic costs as an objective in decision making at watershed scales. When water scarcity costs were not included as an objective (single objective maximizing quality-weighted connected habitat), water scarcity losses considered post model run were at least $400 M greater than when water scarcity costs were minimized (Figure 13).

Diversions and road crossings were the most frequently removed barriers when considering all months and weights (Figure 20) even though road crossings make up about 24% of the total barriers. Road crossings were, on average, cheaper than other barriers but their removal recurrence in all budget levels and objective weights, suggests they play a key role in improving habitat connectivity. Diversions were also commonly removed, likely because of the high number of diversions in the basin (Figure 23).

Incorporating fish passage through barriers as a penalty highlighted barriers that inhibit fish movement. When fish passage was not included in the model, at a $1.5 M budget, 42% (5 out of twelve) of removed barriers were rated as mostly or fully passable when subsequently considered. Including barrier passage as a penalty, 30% (3/10) of removed barriers were mostly passable and the model did not remove any fully passable barriers. While removal of fully passable barriers may help restore a stream to its natural
state, removal is not necessary to increase fish habitat connectivity. However, a fully passable barrier could still negatively alter habitat conditions. The passage penalty may not be ideal if improving habitat was preferred over fish movement in the watershed.

When a metric of habitat connectivity was not included in model formulation, total habitat gain was unaffected, but suitable habitats were spread across watersheds, instead of habitats centered together (Table 13). If connected habitats are not valued for river restoration, habitat gains could come at an average of $21 M less (for barrier removal budgets less than $50 M). With restoration budgets over $50 M, habitat could be gained at a similar price (499 km per $100 M, Figure 18). Removing the habitat connectivity index indicates the importance of considering habitat connectivity. If reaches remain fragmented or inaccessible, habitat gains may not benefit species with large ranges, like Bonneville cutthroat trout.

The longest single reach length was greatest (286 km at $80 M budget) when quality-weighted habitat was favored over minimizing water scarcity, but water scarcity was also greater compared to other objective weights. The average reach length increased as the objective weight on water scarcity losses increased. As the weight increased, favoring minimizing water scarcity costs, the large, economically important barriers were not removed, fragmenting reaches. Rather than one single large reach, the model favored grouping barrier removals, creating numerous connected reaches (average reach 22 km at equal weights). This is important to consider because if the management goal includes removing all barriers from a reach or area, there may be a limit to the maximum reach length. Additionally, focusing barrier removal within one area rather than spreading
efforts throughout the entire watershed will help improve habitat connectivity to maintain sub-populations of fish.

Ideal reach lengths also influence location and composition of removed barrier sets. A watershed manager might look at the entire basin to determine barrier removal, but the connectivity and improvement of specific reaches may occur at a smaller scale. Ability to adjust ideal connectivity thresholds is a useful tool to cater model results for specific goals, species, or assemblages. For example, if modeling for a non-migratory species, it may be favorable to limit barrier removals within a suitable habitat range. This analysis focused on results from the entire watershed area and 30 km reach lengths because it is sufficient habitat range for Bonneville cutthroat trout at all life stages (Carlson & Rahel 2010; Colyer et al. 2005; Young 2011).

Model inputs and soft penalties can be adjusted depending on the watershed network constraining criteria or management questions. The ability to adjust model penalties and input allows flexibility to apply the barrier optimization to different watershed networks and fish species. For example, changing input habitat suitability criteria for another fish species will produce a different set of results. Removing specific barriers from decision making due to their environmental benefit, or infeasibility to be removed, allows the model to prioritize barrier removal while considering permanent barriers. Keeping some barriers in place could be a tool for decision-makers to adjust the model based on local knowledge. For example, if a barrier was in place to block invasive species from a stream reach, it might be desirable to exclude the barrier as a removal option.
During different times of the year, changing environmental conditions limit habitat quality. Adjusting suitable aquatic habitat conditions changes prioritized barrier removals. In summer months, the primary constraint to suitable habitat is discharge and temperature, while in spring months the main environmental inhibitors are gradient and geomorphic conditions. Differences in barriers removed between months changed slightly, but barriers were removed repeatedly in many model runs (Figure 27). Assessing which physical and water quality attributes limit habitat is important for restoring 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.

Limiting number of barrier removal projects and/or capping the budget and economic loss produce different results. For example, by restricting number of projects to one and economic losses to zero, the model removed one of three dams depending on objective weights and month (Figure 26). Increasing the project limit to five barriers, created a different set of barrier removals because the model considered the additional constraints (Figure 26). Although limiting habitat characteristics differ by month, this exercise is useful to quickly identify a subset of the most promising barriers to remove.

My optimization model is one of a few models or tools available to understand modern, multi-objective, and complex decisions. This model was developed as an academic modeling exercise but model results and utility were communicated with local watershed managers. Recommended model input data and scenarios were incorporated into the model, and future work will include expanding the application within the Weber Basin and other watersheds.

5.1 Limitations

Data availability, numerous assumptions and simplifications of reality limited the modeling and data development process. Weber River barrier removal systems analysis included only natural, perennial rivers. Canals, ditches and small intermittent streams were assumed not to have conditions suitable for fish habitat and were not included in the analysis. Barrier passage ratings for Bonneville cutthroat trout are assumed constant
FIGURE 27. Promising barriers to remove, highlighting the inhibiting aquatic habitat condition at each barrier throughout the year and for fish moving upstream or downstream. In future work expanding barrier passage ratings and cumulative passability could be included with available data or knowledge of barrier passabilities. Also, barriers were either removed or not removed and alternative fish passage options were not considered.

Changing barrier passage with hydrologic conditions could increase the number of passable barriers in spring months and decrease the passable barriers in summer months. Large, more harmful barriers may be chosen over other barriers during all times of the year. Cumulative passability might change barrier removal sets by favoring barriers directly adjacent to each other and located in one area of the watershed. For example, in the $1.5 M (Figure 27) scenario, barriers were primarily removed in two
different areas. Cumulative passage would benefit habitat by removing barriers in one area of the watershed. Considering alternative restoration options such as fish ladders may increase the probability of choosing more expensive and large barriers.

I modeled each month at different reach connectivity lengths and my analysis focused on long reach lengths and the entire watershed. In winter months a large habitat range might not be necessary for fish but in spring months a larger habitat range may be crucial to provide access spawning habitat. It may be more appropriate to focus on small reach lengths when considering improving winter habitat.

I assumed that 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 not considered, although monthly variability is considered. I did not consider all age and size ranges of a fish. A young-of-the-year fish has different habitat requirements than an adult fish. The only economic water use considered was water supply from ten barriers, although hydropower, flood protection, and recreational benefits could be added to future models.

The index of connectivity does not include barrier fish passage, rather assumes each reach or habitat above a barrier is inaccessible to adjacent reaches. This is important because while the index measures connectivity, it may be an underestimate. This model is a tool for decision makers and does not replace expert knowledge and judgment.

My model maximized total length of suitable habitat. A reach with barriers in tributaries or located in one single stream segment were treated equally. Reaches including multiple tributaries can provide diverse habitat and may be preferred over a
single length of stream. To test the idea of giving preference to reaches containing multiple tributaries, barriers reconnecting tributaries were given a greater emphasis over barriers in a single length of river. For example, two unimpeded barriers located in separate river tributaries were given preference over two unimpeded barriers located in the same single river length. Initial model runs favoring tributary reaches did not differ from barrier removal sets where tributary reaches and single reaches were treated equally. In future work it would be beneficial to address barrier removal set structure and habitat characteristic variability.

As large barriers were removed, reach habitat quality changes. For example, if a large dam is removed, habitat suitability downstream of the dam will change. This dynamic habitat change was not accounted for as barriers were removed. Removing large dams will return rivers to a natural flow regime (Poff et al. 1997), but large reservoirs can be operated to maintain cold stream temperatures, beneficial to downstream cold water fish populations (Rheinheimer et al. 2015).

Costs of barrier removals were estimated and generalized to illustrate barrier removal options. Sensitivity analysis of barrier removal cost estimates change the slope and inflection point of habitat and cost tradeoffs. I recognize that every project cost differs depending on numerous conditions. Improving barrier removal cost estimates is a needed direction for future research.
CHAPTER VI
CONCLUSIONS

This study prioritized barrier removal using dual objective optimization to maximize quality-weighted, connected habitat and minimize water scarcity costs of reduced water deliveries to cities and agriculture. The model incorporates habitat suitability created from discharge, water temperature, gradient and geomorphic conditions. An index of habitat connectivity estimated each barrier’s centrality and contribution to habitat connectivity. Ability of Bonneville cutthroat trout to move beyond a barrier established a barrier’s passability penalty, where impassable barriers receive a greater penalty. Economic losses due to lost water deliveries were considered for seven reservoirs and three diversions. A budget for barrier removals constrained the model. Barrier removal costs from the American Rivers database and expert opinion estimated removal costs of diversions, impoundments, and road crossings. Results were visualized as Pareto-optimal tradeoff curves, where each value on the curve represents a different barrier removal set. Tradeoff curves of habitat gain versus economic losses and removal costs provide a set of optimal solutions for decisions makers to evaluate with expert knowledge.

This analysis produced five main conclusions that illustrate the advantages of optimization modeling and from the barrier removal case study in the Weber Basin. First, there are diminishing returns in investment as more barriers are removed. The initial $10 M spent on removing barriers to connected suitable habitat produced more habitat per dollar than the last $10 M. Understanding habitat gains over a range of restoration
budgets for barrier removal is beneficial for watershed managers to make efficient management decisions. Second, removing numerous small barriers produced the same or more habitat with lower economic losses from lost water deliveries, compared to removing large barriers. Economically beneficial barriers were not removed until high budget levels and resulted in less cumulative habitat gained. Road crossings were the most frequent barriers chosen for removal across all budget levels and objective weights indicating their importance in habitat connectivity and fragmentation in the Weber watershed. Third, water scarcity costs were important to consider as a second model objective. Without accounting for water scarcity, water scarcity losses increased compared to the dual objective model. Fourth, model results change depending on management preferences and questions. Results changed depending on input criteria and constraints, demonstrating the benefit of an optimization approach. When modeling smaller habitat connectivity thresholds of Bonneville cutthroat trout, the model prioritized barrier removal in tributary reaches. When habitat suitability was optimized without a connectivity index, habitat gain increased slightly but habitat gain was not necessarily accessible or connected. Fifth, optimization modeling is a promising approach to consider both human (economic) and environmental objectives in river restoration and water resources management. The optimization model successfully incorporated numerous objectives and habitat criteria to determine the most appropriate solutions given the conditions.

This modeling approach was demonstrated with a case study in the Weber River watershed, although the optimization model is generalizable to other systems (Brown et al. 2015). When considering multiple objectives with constraints for hundreds of barriers,
removal decisions are complicated. An optimization approach offers a feasible method to consider multiple objectives of connecting habitat and maintaining water deliveries at the landscape scale. 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.


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APPENDICES
APPENDIX A:

NOTATION
<table>
<thead>
<tr>
<th>Set/Parameter</th>
<th>Definition</th>
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<tbody>
<tr>
<td>IIC</td>
<td>The landscape degree of habitat connectivity</td>
</tr>
<tr>
<td>$a_i$ and $a_j$</td>
<td>The habitat area of stream reaches $i$ and $j$</td>
</tr>
<tr>
<td>$A_L$</td>
<td>Total stream reach length</td>
</tr>
<tr>
<td>$P_i$ and $P_j$</td>
<td>Penalty on barrier $j$ and $i$, based off passability of barrier $j$ and $i$</td>
</tr>
<tr>
<td>$B_k$</td>
<td>If a barrier is removed (1) or not (0)</td>
</tr>
<tr>
<td>$c_k$</td>
<td>Water scarcity costs between barriers $i$ and $j$</td>
</tr>
<tr>
<td>$w$</td>
<td>Objective weight</td>
</tr>
<tr>
<td>$h_k$</td>
<td>Habitat suitability upstream of barrier $k$</td>
</tr>
<tr>
<td>$G_k$</td>
<td>Reach gradient upstream of barrier $k$</td>
</tr>
<tr>
<td>$T_k$</td>
<td>Water temperature upstream of barrier $k$</td>
</tr>
<tr>
<td>$G_{C_k}$</td>
<td>Geomorphic condition upstream of barrier $k$</td>
</tr>
<tr>
<td>TC</td>
<td>Total budget for barrier removal</td>
</tr>
<tr>
<td>$C_k$</td>
<td>Cost of removing barrier $k$</td>
</tr>
<tr>
<td>$H_{ql_k}$</td>
<td>Habitat score upstream of barrier $k$</td>
</tr>
<tr>
<td>$H_{L_k}$</td>
<td>Reach length upstream of barrier $k$</td>
</tr>
<tr>
<td>$H_{w_k}$</td>
<td>Quality-weighted habitat upstream of barrier $k$</td>
</tr>
<tr>
<td>CL</td>
<td>Culvert Length</td>
</tr>
<tr>
<td>$T_{A_{a,k}}$</td>
<td>Air temperature</td>
</tr>
<tr>
<td>$t$</td>
<td>Month</td>
</tr>
<tr>
<td>$C_{R_{ij}}$</td>
<td>Binary decision of reconnecting habitat between $i$ and $j$</td>
</tr>
<tr>
<td>$I_{nt_{i,j,k}}$</td>
<td>The sum of barriers located between $i$ and $j$</td>
</tr>
<tr>
<td>$L_{i,j}$</td>
<td>Topological distance between $i$ and $j$</td>
</tr>
</tbody>
</table>
APPENDIX B:

HABITAT SUITABILITY MAPS
FIGURE B-1. January habitat suitability

FIGURE B-2. February habitat suitability
FIGURE B-3. May habitat suitability

FIGURE B-4. April habitat suitability
FIGURE B- 5. May habitat suitability

FIGURE B- 6. June habitat suitability
FIGURE B- 7. July habitat suitability

FIGURE B- 8. September habitat suitability
FIGURE B-9. October habitat suitability

FIGURE B-10. November habitat suitability
FIGURE B-11 December habitat suitability
APPENDIX C:

HABITAT LIMITING MAPS
FIGURE C-1. January limiting habitat criteria.

FIGURE C-2. February limiting habitat criteria.
FIGURE C-3. March limiting habitat criteria.

FIGURE C-4. April limiting habitat criteria.
FIGURE C- 5. May limiting habitat criteria.

FIGURE C- 6. June limiting habitat criteria.
FIGURE C- 7. July limiting habitat criteria.

FIGURE C- 8. September limiting habitat criteria.
FIGURE C- 9. October limiting habitat criteria.

FIGURE C- 10. November limiting habitat criteria.
FIGURE C-11. December limiting habitat criteria.