Instream Flows: New Tools to Quantify Water Quality Conditions for Returning Adult Chinook Salmon

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Abstract: This paper examines the effect of implementing a water transaction program to address potential water quality limitations for returning adult fall-run Chinook salmon in a stream system where the agriculture is the dominant land and water use. Water transactions are becoming an increasingly used approach to provide instream flows during periods when there are competing water uses. Water transactions are often used to achieve ecological objectives, but their water quality or biological effects are rarely quantified. The effects of a water transaction implemented in the Shasta River were evaluated by using a spreadsheet model to quantify changes in dissolved oxygen conditions as they relate to discharge, pool volumes, holding habitat capacity, and potential dissolved oxygen demand by holding fish. The results indicate that water transactions may mitigate potential water quality impairments by decreasing the residence time in holding habitat, and are particularly effective during periods when flows are low, holding habitats are near carrying capacity, and dissolved oxygen demand by fish is elevated. **DOI: 10.1061/(ASCE)WR.1943-5452.0000590.**

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Introduction

Returning adult fall-run Chinook salmon (*Oncorhynchus tshawytscha*) face potential obstacles in basins where their natal streams have reduced flows due to land use and associated water demands (Sheer and Steel 2006; Waples et al. 2009; Keefer et al. 2010). Specifically, where irrigation demands coincide with aquatic habitat needs, water shortages may result for both demands, particularly during dry years (Downard and Endter-Wada 2013). Whereas seasonal low-flow conditions and periodic drought are natural occurrences in freshwater ecosystems, extended drought-like conditions can seriously impair ecosystem function (Gasith and Resh 1999; Lake 2003; Lindley et al. 2007; Arthington et al. 2010).

Securing water for environmental outcomes is currently gaining momentum in the United States and in Australia and South Africa (Postel and Richter 2003). Flow restoration programs based on environmental flow prescriptions are currently under way in many basins in the United States, including the Colorado River in the Grand Canyon (Meretsky et al. 2000), the Trinity River in California (Stalnaker and Wick 2000), and the Kissimmee River in Florida (Toth 1995). Large-scale, multiagency flow restoration programs such as these illustrate a generally top-down approach to managing watershed-scale resources that involve coordination or reoperation of large infrastructure (Hillman and Brierley 2005). Despite these well-known efforts, in general, very few environmental flow programs have been implemented, and even fewer in a research and management context (Davies et al. 2013). Davies et al. (2013) noted that society's willingness to allocate water to sustain the environment is increasingly competitive. Where environmental flow programs have been proposed or developed, several challenges have been identified, particularly with regard to transaction quantification.

New types of environmental flow programs aimed at securing water voluntarily from private water rights holders are currently gaining momentum in the West. In the past decade, environmental flow programs such as the Columbia Basin Water Transaction Program and Scott River Water Trust have been created to provide water instream while working collaboratively with willing water right holders (Garrick et al. 2009; Garrick and Aylward 2012; Wheeler et al. 2013). These programs follow the broader trend of providing environmental flows that have been designed to focus on fish species with significant recreational or commercial value, particularly salmonids in the cool-water streams of North America (Jensen and Johnson 1999; Tharme 2003). Cooperative agreements with water right holders that augment instream flows are also being negotiated and implemented in the Shasta Basin and are the subject of this study.

The problem of securing water supply for environmental purposes is twofold. First, in much of the West, conservation and regulatory policies focus on aspects of aquatic ecosystems (such as water temperature and sediment) rather than developing mechanisms to secure the water supplies that are crucial to ecosystem function (Soule et al. 2005; Fisher et al. 2008). Secondly, environmental objectives may not be considered a "beneficial use" of a water right.

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Because environmental water uses were not legally recognized by most states until the late 1970s, water rights for many of these uses have far less secure, more "junior" water rights (Getches 2009; Hillman et al. 2012; Downard and Endter-Wada 2013). Therefore, securing water supplies for environmental objectives may be challenging given the existing "first in time, first in right" water rights framework that is commonly implemented throughout the arid West (Downard and Endter-Wada 2013). Alternative approaches are providing flexibility to access water supplies independent of seniority-based frameworks (Baron et al. 2002; Grey and Sadoff 2007). During times of drought, cooperation between neighboring landowners outside the formal requirements of water law can alleviate negative impacts of water scarcity (Endter-Wada et al. 2009).

Despite the lack of environmental flow augmentation programs, researchers have advanced a variety of methods to develop flow prescriptions aimed at improving ecological health over the past two decades (Postel and Richter 2003). Many methods primarily focus on water quantity without explicitly considering water quality (Olden and Naiman 2010; Null and Lund 2012). Nonflow, spatiotemporal variables such as water quality (Jager and Smith 2008; Null et al. 2010) and geomorphology (Null and Lund 2012) have been identified as critical to identifying effective water management actions with environmental objectives. When nonflow variables are considered, another challenge is trying to translate ecological principles into hydrologic management recommendations for particular river basins (Poff et al. 2003). For example, approaches such as DRIFT (King et al. 2003) and ELOHA (Poff et al. 2009) have been developed to identify environmental flow regimes; however, these approaches involve substantial resources to quantify variables across multiple disciplines (King et al. 2003; Pahl-Wostl 2007) or assess the water system at global (Pahl-Wostl 2007) or regional scales (Poff et al. 2009). When environmental flow experiments are developed to address specific questions in particular reaches, critical information is provided but rarely within short management timeframes (Arthington et al. 2006). Regardless of the timing, scale, or recommended breadth of human, physical, or biological variables, effective environmental flow recommendations include two factors: the spatiotemporal variability of water quality conditions and their relationship to ecological function in targeted stream reaches.

The objective of this study was to develop a method to quantify the effects of flow transactions on physical habitat and water quality. Specifically, this study focused on the relationship between stream flows, habitat capacity, and dissolved oxygen conditions in holding habitat for adult fall-run Chinook in the Shasta River. The results of this study help quantify the direct benefit of individual, small-scale flow transactions on ecological conditions. Also, the findings help improve understanding of the processes through which these benefits are provided. The tools and methods developed during this study can be broadly applied to basins in which environmental flows have been proposed to improve ecological conditions by providing quantification and transparency to the flow transaction process.

Study Basin

The Shasta River is a tributary to the Klamath River located in Siskiyou County, California. The stream supports an array of anadromous salmonids, including fall-run Chinook and coho salmon, and steelhead trout. Agricultural and municipal water resources development has occurred over the past century. In this highly managed, spring-fed and snowmelt-runoff basin, low flows during the end of the irrigation season (April 1 through September 30) result from a combination of minimal seasonal base flows and water diversions (Nichols et al. 2014). Prior to recent restoration activities and changes to water management practices in the Shasta Basin, decades of poor grazing practices and water diversions degraded water quality and physical habitat conditions for multiple life stages, including migration (MTI and HSU 2014). As a result, the estimated returns of Southern Oregon-Northern California Coastal Evolutionary Significant Unit (SONCC-ESU) adult fall-run Chinook salmon (FRC) have declined from over peak numbers in the early twentieth century of 80,000 to generally less than 10,000 in more recent years.

Although FRC in the SONCC-ESU are not listed as threatened, Chinook salmon generally have considerable commercial and recreational value in California (Yoshiyama et al. 1998; Weise and Harvey 2005); in 2013, Klamath fall-run Chinook accounted for approximately 29% of California's commercial and recreational fishery (PFMC 2014). It is estimated that 30% of Klamath River salmonids originated in the Shasta River (Moyle et al. 2008). Because of voluminous spring-fed sources that provide desirable physical and water quality conditions for salmonids, and support a robust food web, the Shasta River is the focus of extensive conservation activities that focus on restoring and maintaining the aquatic ecosystem (Jeffres et al. 2010; Willis et al. 2012).

Throughout the basin, water resource users and managers are modifying water use activities to include environmental objectives; however, competing demands remain a challenge. A flow transaction program is one proposed management strategy to add flexibility to the existing water right allocation system that defines water use in the Shasta Basin. Water rights in the Shasta Basin are a mix of riparian and appropriative water rights that were adjudicated as per the 1932 Court Decree (DWR 1932). The Shasta River Water Transaction Program, developed in 2009, provides a mechanism for willing individual water right holders to contribute water to instream flows for targeted, short-term periods. Generally, these contributions focus on specific spatial, temporal, quantity, and quality objectives.

Although the primary limitation to salmonid recovery in the Shasta Basin is insufficient oversummering habitat caused by elevated water temperature conditions (Jeffres et al. 2008, 2009), agricultural water use also coincides with adult FRC migration. Agricultural diversions that occur during September coincide with adult FRC staging period, when the fish hold in downstream pools (in a reach commonly called the Canyon Reach) before migrating to upstream spawning habitat. Currently, seasonal low flows during this staging period create the potential for impaired holding habitat conditions (Morgan Knechtle, personal communication, 2014). Flows increase after irrigation activities cease on October 1. Within days of the end of seasonal irrigation activities, base flows at the mouth of the Shasta River nearly double, increasing an average of 1.8 m^3/s and largely ameliorating adverse conditions in the Canyon Reach.

Preliminary instream flow recommendations have been developed for the early (September 15–30) adult Chinook migration in the Canyon Reach of the Shasta River but have not been formally implemented (MTI and HSU 2014). Further research into the ecological relationship between stream flows and migratory cues are important for long-term management planning in the Shasta Basin (Willis et al. 2013). However, until those relationships are clarified, an assessment of existing conditions and potential short-term management strategies has been explored through the use of voluntary flow transactions during this critical fall period.

Recent returns of adult FRC to the Shasta River have been well above average. Fish counts of adults entering the Shasta River indicate that, until 2011, the number of returning FRC before

October 1 was 173–3,481 fish (Chesney and Knechtle 2014). The number of FRC that returned to the Shasta River before October 1 increased to more than 5,600 fish in 2011 and more than 17,500 fish returned in late September in 2012. The higher returns have prompted resource managers to consider the implications of potential impairments associated with limited habitat and water quality conditions.

Previous surveys of spring-run Chinook in holding habitat suggest that pools have a limited holding capacity, characterized by an asymptotic relationship between specific pools and number of fish (Campbell and Moyle 1992). High fish densities (i.e., near the carrying capacity of the surrounding water) can rapidly deplete dissolved oxygen, particularly if fish are stressed (Portz et al. 2006) because of crowding, elevated temperatures, limited or suboptimal holding habitat, or other conditions. Given these chronic stressors, fish experience more severe effects (Pickering 1981; Barton 2002) from which they are less likely to recover (Portz et al. 2006; Newcomb and Pierce 2010). Previous studies have identified relationships between water quality conditions experienced by spawning adults and the fitness and development of larval and juvenile offspring (Burt et al. 2011); generally, such studies have focused on water temperature (EPA 2003).

Dissolved oxygen (DO) is another critical, although less studied, component of holding habitat. Although much uncertainty remains around the relationship between low DO and adult Chinook fecundity (Newcomb and Pierce 2010; Strange 2010), minimum DO concentrations of 5 mg/L and at least 80% saturation are generally recommended for migrating adult Chinook (Westers and Pratt 1977; Bjornn and Reiser 1991; Newcomb and Pierce 2010; Strange 2010). Aquaculture experiments indicate that during periods of peak activity by migrating adults in holding ponds, DO concentrations can become less than 6 mg/L, even given inflows near saturation (McLean et al. 1993). Few studies have examined DO concentrations and potential effects on Chinook salmon holding in natural, riverine habitats.

Given the potential impairment suggested by previous studies, and the need to support the ongoing recovery of this sensitive species in the Shasta River, an experiment was designed to quantify potential DO conditions that could result from a combination of larger returns of adult FRC, low water conditions, and spatiotemporal water quality of holding habitat in the Canyon Reach of the Shasta River. This study was designed in two phases to quantify several factors and inform management decisions of cold-water species during a period of competing water use.

Phase 1 occurred in 2012 and tested quantification methods to evaluate impacts of water transactions on physical habitat. Positive outcomes from Phase 1 activities led to Phase 2 activities in 2013. During Phase 2, the Phase 1 quantification methodology was applied, and the analysis was extended to include the water quality elements. By using the data collected during flow transactions that occurred in 2012 and 2013, a methodology was developed and tested to evaluate water transaction effects on physical habitat in a representative pool. A spreadsheet model was developed to quantify the potential DO changes in holding habitat caused by the presence of adult FRC and assess the extrapolation of predicted outcomes to larger study reaches.

The Phase 2 hypothesis was that increasing discharge would mitigate potentially low DO conditions caused by high fish densities through increasing pool volume and decreasing pool water residence time in these principal holding habitat areas. Phase 2 was based on previous studies that identified relationships between adult Chinook, water temperature, and oxygen consumption (respiration) (Farrell et al. 2003; Clark et al. 2008).

The study area included the 13.7 km reach of the Shasta River from its confluence with the Klamath River upstream to the confluence with Yreka Creek, located near the Interstate 5 crossing (Fig. 1). This reach comprises the area where adult FRC hold before the end of irrigation season on October 1. Approximately one-third of the reach contains pools and nontraditional "mesohabitat" designations, including "run-pools" that function as holding habitat types for adult FRC (MTI and HSU 2014).Two pools, Hudson Pool and Weir Pool, were identified to test the quantification methodology. Both pools were located near existing fish counting facilities and monitoring stations providing flow data. Hudson Pool is located approximately 1.5 km upstream from the mouth of the Shasta River and 1.3 km upstream from the Shasta River Fish Counting Facility. Weir Pool is located approximately 90 m upstream from the mouth of the Shasta River and 100 m downstream from the Shasta River Fish Counting Facility.

Methods

Transaction Implementation and Quantification

Transaction objectives and location were determined on the basis of previous field observations of hydrologic and water quality conditions throughout the Shasta River, and life history strategies that indicated adult FRC holding in the Canyon Reach of the Shasta River. The transaction period was determined according to the historic timing of adult FRC returns to the Shasta River and the irrigation schedule for the basin. The timing of the adult FRC returns was determined by using fish count data collected by the California Department of Fish and Wildlife (CDFW) at the Shasta River counting weir. Between 2001 and 2013, the earliest adult FRC was detected at the counting weir on September 1. Seasonal irrigation activities in the Shasta Basin end on September 30. To minimize the disturbance to potential fish at the study sites, the study period was extended to allow the setup of monitoring equipment before the expected arrival of adult FRC and removal of that equipment after their upstream migration from potential holding habitat. Thus, the study period occurred between August 28 and October 2.

Coordination of the water transactions was facilitated by The Nature Conservancy, the Shasta Valley Resource Conservation District, California Department of Fish and Wildlife, and the Scott Valley and Shasta Valley Watermaster District. Outreach to the community occurred in the form of newspaper articles, community meetings, one-on-one phone calls, and e-mails. Contributions of water rights were voluntary and without any formal agreement in place. Each participant was responsible for self-reporting the amount they ceased diverting (i.e., contributed to the water transaction). Other contributions of water were inventoried with assistance from the Watermaster.

Because the transaction focused on conditions in holding habitat (i.e., pools), transactions were quantified developing an aquatic habit rating curve consistent with Gippel and Stewardson (1998). This curve applied geomorphic principles to quantify the amount of aquatic habitat created by specified amounts of discharge added to a specified baseline (i.e., nontransaction) discharge. Pool volume was the metric used to quantify changes to aquatic habitat.

Discharge was monitored by using the USGS stream gauge located in the Shasta River near Yreka, California (11517500), which operates in real time and is publically accessible through both the USGS and California Data Exchange Center websites. Subhourly data was downloaded for the periods of August 28 through October 7, 2012, and August 28 through October 2, 2013.

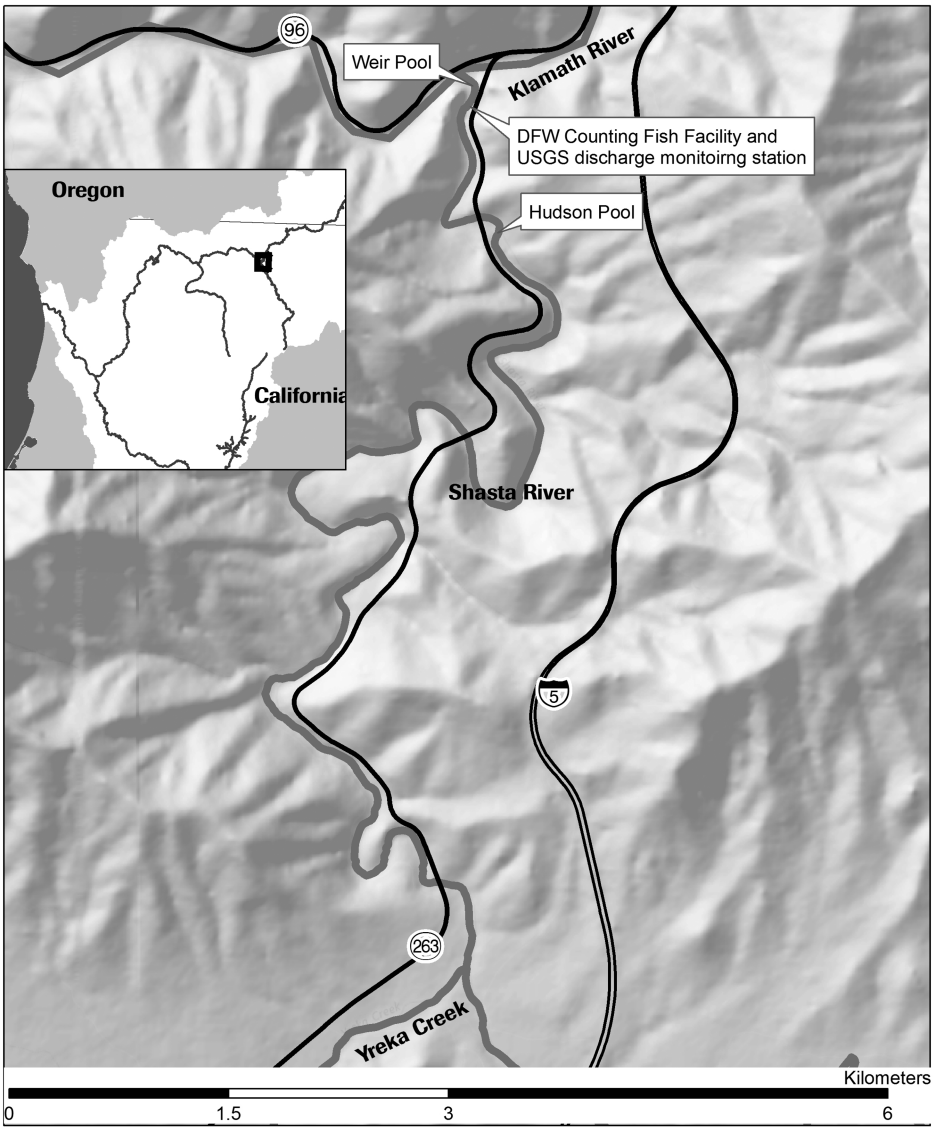

Fig. 1. Canyon reach of the Shasta River and locations of the study pools: Hudson Pool and Weir Pool

Pool volumes were monitored by using methods developed by the National Fish and Wildlife Foundation (NFWF) to relate discharge to physical aquatic habitat (Holmes et al. 2013). Five cross sections (i.e., TA, TB, TC, TD, TE) were established perpendicular to and approximately equidistant along the longitudinal centerline of the stream channel. Bank pins were monumented to establish fixed elevation data at each transect. The location of the upstream pool boundary was identified at the hydraulic transition from an upstream shallow riffle to slow-moving, deeper streamflow (Fig. 2). The downstream boundary was identified by a riffle crest. The pool length was measured as the average distance between pins on the river left and river right banks (direction is relative to the viewer looking downstream). For each transect, initial surveys were completed by using a measuring tape and construction string fixed to the bank pins. Depth measurements were taken at regular intervals along each transect by using a measuring tape and stadia rod. The depth from the tape/twine line to the stream bed and to the water surface were measured at a minimum of 22 points located between the left and right bank pins, including the left edge of water (LEW), right edge of water (REW), and the channel thalweg (deepest part of the channel). The remaining 19 survey points were located at intervals of 5% of the distance between both bank pins (5, 10, 15, . . . ,95%). For measurement points located out of the water, only the distance below the tape/twine line were recorded.

The approach allowed cross-section measurement locations to be reoccupied during subsequent surveys. Low flows supported the assumption that bed materials during the study period were immobile; thus, bed topography would remain unchanged. To minimize any disturbance of fish present during the survey, the twine line remained in place through the study period, and subsequent surveys were completed by noting the water surface elevation and distance between the wetted edge and bank pin. Five total surveys were completed at each cross section. Pool volumes, V_p, were calculated by summing the products of A_{xsn}, the average cross-sectional area of two adjacent surveyed transects, and l, the sum of the centerline distance between each adjacent transect

$$V_p = \Sigma(A_{xsn} \times l) \tag{1}$$

Because the data would be used to develop a discharge-pool volume rating curve, the timing of each survey was determined by changes in discharge to include a wide range of flows.

Fig. 2. Aerial photo of the 2013 study sight, the direction of discharge (Q), and the five transects (TA, TB, TC, TD, TE) that were monitored for physical aquatic habitat changes (image © 2013 Google)

Relating Water Quality and Fish

A spreadsheet fish habitat-DO model was developed to examine the relationships between discharge, water quality, carrying capacity, and oxygen consumption in pool habitats. The framework for this model was designed to focus on key physical habitat changes and oxygen demand processes while representing simplified water quality dynamics, hydraulics, and fish distribution in a pool setting. Key processes included a direct calculation of changes in physical habitat capacity and dissolved oxygen demands by adult FRC given specified discharge and initial water quality (i.e., water temperature, DO) conditions.

Initial concentrations of DO (i.e., DO concentration of instream flows before consumption by adult FRC) in the pool were assumed to be the same as upstream inflow to Weir Pool. Subsequently, the fate of DO given fish respiration was determined for the defined pool and number of fish. Initial DO concentration and water temperature were quantified by using observed data that were logged at a 0.5 h interval using an Onset U26-001 (Onset Computer Corporation, Bourne, Massachusetts) optical logger. This logger measures DO from 0 to 30 mg/L with an accuracy of 0.5 mg/L and water temperature between −5 and 40°C with an accuracy of 0.2°C. The logger was suspended above the stream bed in an open-ended PVC canister to allow ample stream flow through the protective cover. The dissolved oxygen sensor was protected by using an antibiofouling cover and was inspected and serviced weekly to prevent any potential biofouling buildup. The CDFW provided fish count data collected at the Shasta River counting weir, which was used to determine whether numbers of fish present in the system had a potentially substantial effect on observed DO conditions.

Mathematically, the effects of respiration on DO concentration caused by physical habitat capacity and DO demand associated with fish density were represented as

$$\mathrm{DO}_r = (\mathrm{DO}_i - \mathrm{DO}_c)/V_p \tag{2}$$

where DO_r = dissolved oxygen remaining (mg/L); DO_i = dissolved oxygen initial supply (mg); DO_c = dissolved oxygen consumed (mg); and V_p = pool volume (m^3).

DO_i defines DO in the pool before consumption by fish and is calculated as

$$\mathrm{DO}_i = \mathrm{DO} \times Q \times \theta \tag{3}$$

where DO = measured DO in Weir Pool (mg/L); Q = discharge (m^3/s); and θ = residence time (min).

DO_c represents the oxygen consumed by fish that are holding in the pool and is calculated as

$$\mathrm{DO}_c = R \times m \times \theta \times F \tag{4}$$

where R = oxygen consumption rate by an adult Chinook salmon (mg/min/kg); m = mass of an adult Chinook salmon (kg); θ = duration of oxygen supply (equivalent to discharge residence time, min); and F = number of fish.

R is a first-order rate reaction that is calculated as

$$R_n = R_i \times Q_{10}^{(Ti-Tn)/10} \tag{5}$$

where R_n = rate of oxygen consumption at water temperature n (mg/min/kg); R_i = rate of oxygen consumption at temperature i (mg/min/kg); Q_{10} = rate of change of oxygen consumption rate over a 10°C temperature interval, species-specific coefficient; T_i = water temperature at time step i (°C); and T_n = water temperature at time step n (°C).

Clark et al. (2008) specified initial R_i and T_i values, and an empirically derived Q_{10} for an adult FRC ($Q_{10} = 2.2$), which were used to calculate the first iteration of R_n using T_n from data collected during the study period. Subsequent values were calculated by using results from the previous iteration. For all equations, appropriate conversion factors were included as needed.

The mass of an adult Chinook salmon was estimated according to expert opinions of fish biologists studying in the Shasta River. Because this study was designed to consider potential DO impairments caused by fish, conditions that encouraged maximum potential consumption were identified. To evaluate maximum potential demand, the upper limit of approximately 9.8 kg (as suggested by both experts) was used (Carson Jeffres, personal communication, 2013; Morgan Knechtle, personal communication, 2013).

The maximum number of fish holding in a pool (i.e., pool capacity) was calculated according to pool volume. According to the methods developed by Thompson et al. (2012), preliminary estimates of pool capacity were based on surface area (A_s, to estimate

the horizontal distribution of fish) and average depth (d, to estimate the potential layers of fish). Given the relatively shallow depth of Weir Pool, the maximum number of fish was estimated assuming a single layer equal to the A_s of the pool (Lisa Thompson, personal communication, 2013)

$$F = A_s = V_p/d \tag{6}$$

However, multiple layers of fish could readily be accommodated in the model.

During the transaction period, the model assumed that the pool was filled to capacity (i.e., the maximum number of fish). Dissolved oxygen was assumed to be evenly distributed throughout the pool at all times (i.e., homogeneous conditions), and all fish were assumed to have the same respiratory demand (i.e., this version of the model does not account for a range of sizes, therefore consumption rates are fixed). Given the short (i.e., minutes) residence time for each time step of oxygen supply and demand, other potential oxygen supplies (e.g., atmospheric reaeration) and demands (e.g., decay) were considered negligible. When flows and pool capacity decreased, the number of fish represented in the pool decreased to the new maximum capacity; when flows and pool capacity increased, the number of fish increased to the new maximum capacity. These assumptions are consistent with a previous study of dissolved oxygen dynamics during periods of high fish loads in holding habitat environments (McLean et al. 1993).

Results

Transaction Implementation and Quantification

Three irrigation districts and four individual water users participated in the 2012 transaction and cumulatively contributed up to 1.3 m^3/s (Table 1). Lower returning adult FRC counts, delayed in-migration timing, and favorable meteorological conditions during the 2013 transaction reduced the urgency for augmented instream flows. However, The Nature Conservancy (TNC) contributed a portion of its water rights (approximately 0.3 m^3/s) during the 2013 study period for experimental purposes.

Pool volume surveys were completed during both the 2012 and 2013 transactions and were taken across a wide range of flows (Table 2). Discharge in the Shasta River was 0.6–3.1 m^3/s during the 2012 study period and 0.5–4.0 m^3/s during 2013; surveys were completed for discharges of 1.4–3.1 m^3/s in 2012 and 0.5–3.5 m^3/s in 2013. Pool volumes were 1,317–1,714 m^3 in Hudson Pool in 2012 and 276–401 m^3 in Weir Pool in 2013.

Table 1. Summary of the Water Rights Owners and Quantities Contributed to the 2012 Transaction Program

Date	Water right owner	Water right (m^3/s)	Cumulative water added instream (m^3/s)
September 7, 2012	Irrigation district 1	0.17	0.17
September 7, 2012	Individual 1	0.07	0.24
September 10, 2012	TNC	0.27	0.51
September 12, 2012	Individual 2	0.28	0.79
September 14, 2012	TNC	0.24	1.03
September 19, 2012	TNC	0.05	1.08
September 19, 2012	Irrigation district 2	0.23	1.30
September 27, 2012	Irrigation district 1	−0.06	1.24
September 27, 2012	Individual 3	0.06	1.30
September 7–31, 2012	Irrigation district 3	1.13[a]	1.30

[a]Irrigation district 3 (ID3) held a junior water right, permitting it to divert the flows contributed by other transaction participants. ID3 elected not to exercise that right as part of their contribution to the water transaction.

Table 2. Summary of Pool Volumes and Discharge Rates during Transect Surveys in Hudson Pool and Weir Pool

Location	Survey date	Discharge (m^3/s)	Pool volume (m^3)
Hudson Pool	September 4, 2012	1.4	1,317
Hudson Pool	September 14, 2012	1.5	1,343
Hudson Pool	September 25, 2012	2.0	1,471
Hudson Pool	October 9, 2012	3.2	1,714
Weir Pool	August 28, 2013	1.3	331
Weir Pool	September 5, 2013	0.5	276
Weir Pool	September 19, 2013	0.8	298
Weir Pool	September 25, 2013	1.1	320
Weir Pool	October 2, 2013	3.5	401

Aquatic habitat rating curves developed for both Hudson Pool and Weir Pool suggest that the method to assess transactions by relating discharge to habitat metrics provides a robust relationship. R^2 values exceeded 0.999 for Hudson Pool and Weir Pool (Fig. 3). These aquatic habitat rating curves provided a method with which to compare existing conditions to those that would have occurred without the transaction contribution.

The data from Hudson and Weir Pool illustrated that there are a range of pool volume responses to changes in flow, depending on pool morphology. The percent of pool volume created by a 1.3-m^3/s addition to various base-flow rates (i.e., the discharge rate prior to the additional transacted water rate) was calculated for the Hudson Pool and Weir Pool. The results suggest the benefit of flow transactions decrease as the base-flow rate increases. When 1.3 m^3/s was added to 0.1-m^3/s base flow, pool volumes increased 104% in Hudson Pool and 53% in Weir Pool (Fig. 4). However, when 1.3 m^3/s was added to 3.0-m^3/s base flow, pool volumes increased 11% in Hudson Pool and 6% in Weir Pool. These differences also suggest that careful considerations may be required

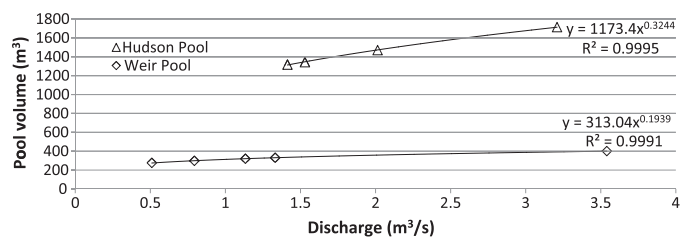

Fig. 3. Discharge-aquatic habitat rating curves developed for the Hudson Pool ($R^2 = 0.9995$) and Weir Pool ($R^2 = 0.9991$)

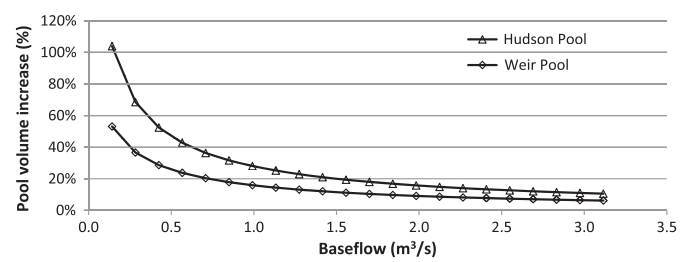

Fig. 4. Percent of pool volume habitat created by a 1.3 m^3/s flow increase given specified base flows. For example, when 1.3 m^3/s is added to a base flow of 0.1 m^3/s, pool volume habitat increases by 104% in Hudson Pool and 53% in Weir Pool

when extrapolating flow transaction effects from a small sample of pools to reach scale.

Water Quality and Fish

The observed fish returns during each transaction period indicate that the timing of the transaction coincides with the beginning of the adult FRC migration into the Shasta River. During the 2012 transaction, adult FRC first entered the Shasta River on September 2, with a notable increase in returns starting on September 6 (Fig. 5). The observed cumulative return before October 1 was approximately 17,500 fish in 2012 and exceeded the average trend by approximately 15,000 fish. During the 2013 transaction, adult FRC first entered the Shasta River on September 9 and continued to enter the tributary at a modest rate until the end of the month. The observed timing and cumulative number of fish reflected average trends. Field observations of the number of fish in Weir Pool were made before transect surveys; only one fish was observed using Weir Pool during the study period. The relatively low number of FRC present in the system during 2013 suggested that the actual oxygen demand by returning Chinook was negligible during the transaction period and that the observed dissolved oxygen conditions were representative of baseline dissolved oxygen in the system. However, the 2012 return suggests that large, early fish returns can occur, leading to high numbers of fish holding in the Canyon Reach. Thus, measured dissolved oxygen may include effects from oxygen demand by fish present in the system.

Observed streamflow, water temperature, and DO data were used as the basis for the initial water quality conditions that would support potential numbers of adult FRC and affect their rate of oxygen consumption. Free-floating vegetative debris was removed from the DO logger during the field visit on September 6, 2013, and data collected before that maintenance were discarded. Streamflows (including the 0.3 m^3/s transaction flow) were 0.5–4.0 m^3/s during the transaction period and were generally less than 1.5 m^3/s until September 26, 2014 (Fig. 6). Water temperatures were 10.3–23.5°C during the study period; the peak temperature was observed on two separate days—September 1 and 13, 2013.

The proposed upper thermal limits to adult Chinook migration in the Klamath Basin are based on the simultaneous occurrence of three conditions: a mean daily temperature of 23°C, a mean weekly temperature of 22°C, and a mean weekly maximum temperature of 23°C (Strange 2010). Whereas mean weekly maximum temperatures exceeded the 23°C threshold, mean daily and mean weekly water temperatures both peaked at 21.3°C, below the proposed thresholds. The DO concentrations were 7.3–11.2 mg/L; the percent saturation was 88–135% (Fig. 7). The DO concentrations and percent saturation both exceeded the recommended minimums of 5 mg/L and 80%. Thus, before consumption by holding FRC, influent DO was satisfactory for holding fish.

Using the observed pool volume and DO data from the 2013 transaction, the fish habitat model was used to quantify potential fish capacity and associated DO conditions (i.e., DO remaining in the pool once the oxygen demand by potentially holding fish was taken into account) with and without the water transaction (the "w/ transaction" scenario and "w/o transaction" scenario, respectively). Although potential fish capacity generally increased with the water transaction (Fig. 8), the transaction had a negligible effect on water quality conditions when holding habitat was at carrying capacity with one notable exception—when water quality conditions were least desirable (i.e., baseline DO was at or below critical thresholds), the transaction had a substantial effect (Fig. 9). From approximately September 10 through 16, 2013, DO conditions with the transaction were notably improved from the condition without transaction (Fig. 9).

A closer examination of the September 9–16, 2013, period illustrates the potential benefit of this water transaction on short-term, adverse conditions. In the w/o transaction scenario, the maximum number of fish that could use Weir Pool as holding habitat was 363–465 adult FRC, with an average of 408 fish (Fig. 10). However, with a 0.3-m^3/s transaction, the pool capacity increased to a maximum of 431–496 adult FRC, with an average of 457 fish.

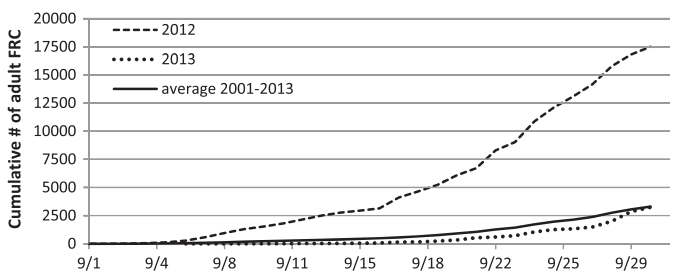

Fig. 5. Summary of the cumulative number of adult FRC that were observed at the Shasta River counting facility

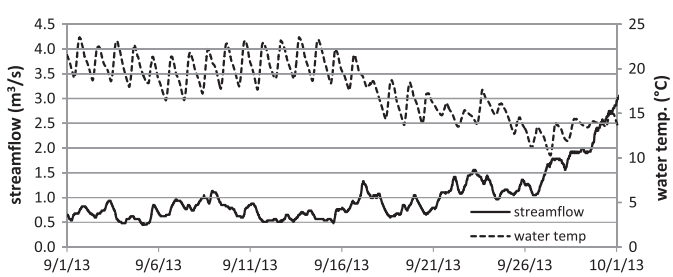

Fig. 6. Streamflows and water temperatures observed in Weir Pool during the 2013 study period

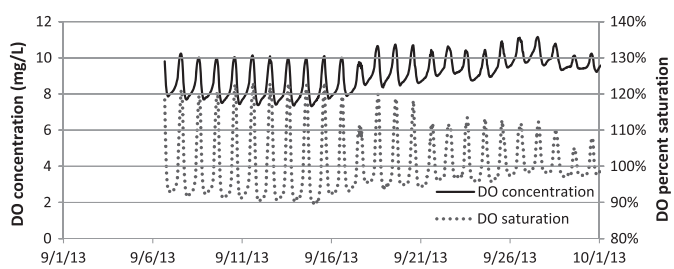

Fig. 7. DO concentrations and percent saturation observed in Weir Pool during the 2013 study period

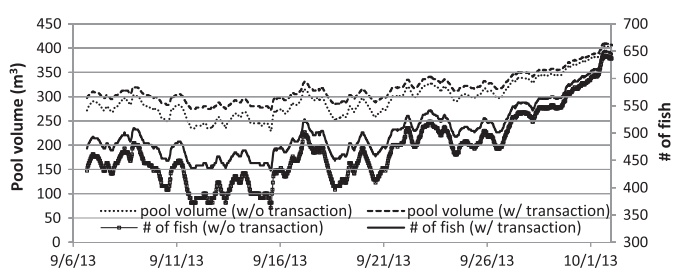

Fig. 8. Summary of pool volume and carrying capacity with and without the transaction; "w/ transaction" condition reflects a 0.3 m^3/s increase in discharge

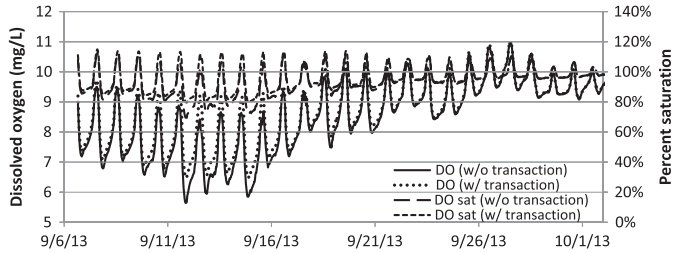

Fig. 9. Summary of DO concentration (mg/L) and percent saturation with and without the transaction; "w/ transaction" condition reflects a $0.3\,m^3/s$ increase in discharge

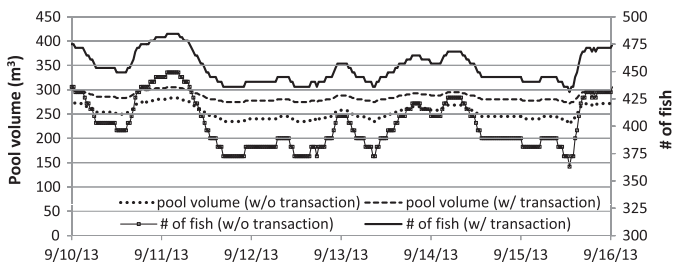

Fig. 10. Changes in observed pool volume and potential fish capacity given a $0.3\,m^3/s$ transaction

The transaction had a relatively greater effect on the minimum number of fish that could be accommodated (68 additional fish) than on the maximum number (31 additional fish). This finding suggests that water transactions provide greater value when used to augment lower base flows.

Aside from accommodating a greater number of fish in the pool, improved water quality conditions are also maintained by the water transaction during this period. Without the transaction, DO concentration and percent saturation were 5.6–9.4 mg/L (average of 7.3 mg/L) and 69–111%, respectively (Fig. 11). With the addition of $0.3\ m^3/s$ to base flows, DO concentration and percent saturation were 6.5–9.5 mg/L (average of 7.7 mg/L) and 80–113%, maintaining DO concentrations well above the recommended 5.0 mg/L and 80% threshold for adults. On average, DO concentration increased with the flow transaction by approximately 0.4 mg/L, with the greatest improvement of 0.9 mg/L from the lowest DO (w/o transaction) concentration on September 11, 2013. The average percent saturation increased by 6%, with the greatest improvement of 11% from the lowest percent saturation (w/o transaction) on September 11, 2013.

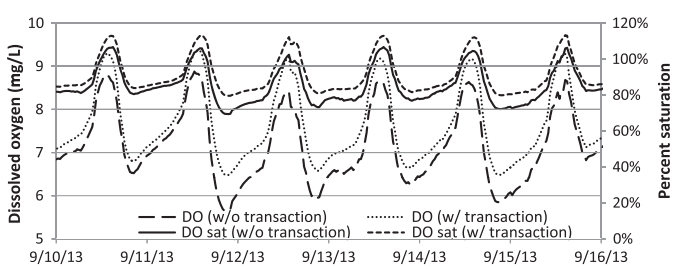

Fig. 11. Summary of DO concentration (mg/L) and percent saturation, with and without the transaction, during the period the transaction had the greatest effect on water quality conditions; "w/ transaction" condition reflects a $0.3\,m^3/s$ increase in discharge

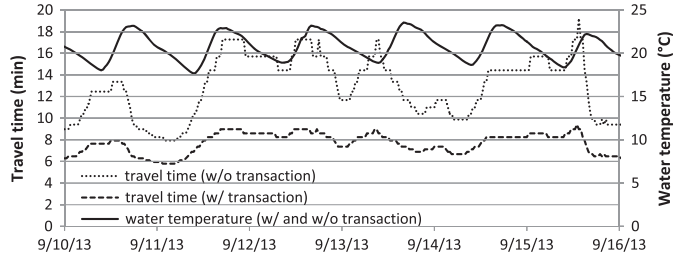

Fig. 12. Travel times and water temperature (both with and without transaction) for Weir Pool given an additional $0.3\ m^3/s$ to base flows that occurred in 2013

Although DO concentrations are modestly sensitive to water temperature (Bowie et al. 1985), an examination of water temperature during the flow transaction indicates that the improved DO conditions are not related to changes in water temperature. Canyon Reach water temperatures are generally at equilibrium with ambient meteorological conditions and show no significant changes for the range of flows assessed in this paper (Nichols et al. 2014). Thus, the fish habitat model assumes that water temperatures are unaffected by relatively modest changes in discharge.

In contrast, travel time of water varies with discharge in the fish habitat model. Given an additional $0.3\ m^3/s$, travel time through Weir Pool is substantially reduced. Without the transaction, travel time through Weir Pool was approximately 8–19 min (based on variable base-flow conditions) during the period when low DO was observed (Fig. 12). With the transaction, travel time through the pool decreased to approximately 6–9 min. The reduced travel time similarly reduces the time during which fish in the pool can consume oxygen from parcels of water flowing through the pool; that is, re-oxygenated water is supplied to the pool (and deoxygenated water is transported out of the pool) more quickly with increased flows.

Discussion

The two-phase water transaction experiment presented an excellent opportunity to explore multiple elements of an environmental flow management action, from its conception and implementation to its quantification and assessment. The two most important contributions made by this experiment are the potential for flexibility in highly regulated basins and quantification of the water transaction benefit that directly relates the water transaction to a potential biological outcome. This approach also demonstrates the successful integration of spatiotemporal variability of water quantity and quality with a specified water management activity.

The implications of the transaction implementation and quantification methodology are extensive. The transaction implementation illustrated that a water transaction framework is a feasible and effective management strategy to provide environmental flows during a period of conflicting demands. The success of this management strategy provides flexibility to secure wet water during critical periods while functioning within the framework of adjudicated water rights. The voluntary participation by water rights holders also suggests that there may be potential to expand this program to include incentives to landowners.

The quantification method is a novel application of geomorphic principles to aquatic habitat evaluation (Nichols et al. 2013; Holmes et al. 2013). Whereas analytical techniques based on hydraulic geometry data are commonly used to develop minimum instream flows (e.g., Jowett 1997; Gippel and Stewardson 1998;

Reinfelds et al. 2004), in this experiment, hydraulic geometry data were more generally used to quantify habitat capacity, travel time, and potential water quality conditions through discharge-aquatic habitat relationships. The strength of the discharge-pool volume relationship was tested in two experimental setting and yielded consistent results. Robust R^2 values (exceeding 0.999 for both Hudson Pool and Weir Pool) suggest that the methodology can be applied over the entire period of holding habitat use for specified locations. However, the actual relationships vary substantially between pools, suggesting that using a single representative pool as a surrogate for assessing reach-scale habitat may be inappropriate. The location and duration of a transaction, and channel geometry, also may affect the application of aquatic habitat rating curves. Alternative approaches may better characterize the relationship between discharge and aquatic habitat in some systems, particularly if alternative parameters (e.g., wetted perimeter) are selected to characterize aquatic habitat (Gippel and Stewardson 1998). Also, short-duration water transactions (several days to weeks) typically do not enable the generation of habitat rating curves using hydraulic rating methods, largely because of an inability to measure habitat variables across a sufficient range of flows.

The transaction quantification methodology also illustrates a cost-effective approach to assess transactions. In the absence of channel-changing flows, the same rating curve may be applicable from year to year, reducing the resources needed to assess available habitat and increasing the value of the methodology. Thus, whereas initially, regular field surveys are needed to develop the aquatic habitat curve, monitoring during subsequent years is simplified to monitoring discharge, water temperature, and dissolved oxygen in the study reach. New surveys would be recommended following channel-changing flows.

Finally, the fish habitat model provides a flexible, transparent method to assess the value of transaction flows. The approach described in this study illustrates how water transactions may be developed and quantified for environmental objectives. Discharge is directly related to potential biological benefits; this relationship provides transparency to natural resource managers and water rights holders. The fish habitat model can be applied to examine potential changes to discharge, water temperature, or return counts, allowing resource managers to assess whether alternative conditions may benefit from a water transaction. By applying the fish habitat model, stakeholders can consider the direct benefit of a proposed transaction flow and make an informed decision regarding participation in a transaction and valuation of the contributed water.

Summary and Conclusions

The key findings of the Shasta River water transaction study include

- Benefits provided by the water transaction are a combination of improved water quality and increased fish capacity in holding habitat;
- Benefits to water quality are a result of reduced travel times and are unrelated to water temperature changes; and
- Benefits from the transaction vary depending on base flow, water quality conditions, and number of fish present in the system.

When evaluating holding habitat conditions, both water quantity and water quality are critical components. This study illustrates that given adequate water quality of influent water, potential impairments were mitigated by simply flushing the holding habitat to more rapidly replace water in the pool. Another notable finding is that not only were water quality conditions improved with the transaction, but more fish could be accommodated without offsetting the benefit from the transaction. When remaining dissolved oxygen conditions improved with the transaction, the "w/ transaction" simulation also included the additional fish that could hold in the pool given the additional habitat created by the transaction flow. Finally, the results illustrate that the benefits of the transaction varied throughout the study periods and generally had little to no effect on water quality conditions when optimal influent water quality conditions were met.

The results highlight the importance of determining life-stage-specific water quality thresholds, as outcomes vary considerably depending on the initial conditions. The fish habitat model helps identify these critical thresholds by linking physical aquatic habitat conditions to biological processes. For the objective of mitigating for high oxygen consumption by holding fish, the critical factors seem to be low flows, partially suboptimal water temperatures, and high returns that would put the pools at carrying capacity. Suboptimal conditions occurred only when previously identified water temperature thresholds were met (mean weekly maximum water temperatures of >23°C). Notably, although the water temperature criteria included three critical thresholds, only one was exceeded to prompt high rates of oxygen consumption, resulting in near-critical DO conditions in the holding habitat. When these conditions are present, the transaction is an effective management strategy. However, as long as ambient water quality conditions met all recommended criteria, the remaining DO concentrations and percent saturation were above recommended levels. Refinements to relationships between water quantity, water quality, and biotic response would improve understanding of critical thresholds that could be used as criteria to implement transactions. Better estimates of holding habitat carrying capacity would also improve the quantification of transaction effects.

The fish habitat model provides a tool with which potential transaction participants (water rights holders or resource managers) directly assess the relationship between a transaction flow and water quality objectives. The model provides transparency for transaction participants, who can use the results to make informed decisions regarding participation in a transaction and the quantified benefit of contributed water. The basic data needs mean that the model can be implemented in any holding habitat setting, given appropriate geometry, discharge, water temperature, and dissolved oxygen data. It can also be applied to help manage other species, given appropriate oxygen consumption coefficients.

Potential long-term benefits include the continued recovery of FRC in the Shasta basin through the reduced exposure to suboptimal DO concentrations, limiting the cumulative effects of long-term stress. Studies suggest that wild adult salmon can make a full recovery from brief periods of moderately elevated oxygen consumption (Farrell et al. 2003); fecundity, offspring fitness, and other factors may benefit when potentially acute holding habitat conditions are moderated through water transactions. Ultimately, the fish habitat model provides a mechanism to balance resource use during periods of conflicting demands in a way that is transparent and quantified. This benefits water rights holders, who can maintain their existing water rights and cooperate with environmental resource managers within a flexible framework, and also benefits fish, for whom potentially stressful conditions can be addressed and mitigated by using targeted management actions.

Additional work includes consideration of the effects of a water transaction given more acute stressors, such as crowding or disease. The results of this study illustrated that the water transaction's benefit varied with other physical and biological factors, and was negligible given desirable influent water quality. Similarly, the water transaction's potential ability to mitigate for more acute conditions

is unclear. Also, more information is needed to improve the scalability of this method. Whereas the aquatic habitat rating curves are robust for targeted areas, an improved understanding of reach-scale carrying capacity would help quantify the reach-scale effect of the transaction.

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