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# An ecohydraulics virtual watershed: Integrating physical and biological variables to quantify aquatic habitat quality

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## Abstract

Advances in remote sensing coupled with numerical modelling allow us to build a "virtual ecohydraulics watershed" at the micro-habitat scale. This approach is an integrated modelling framework with a cascade of models including physical (hydrologic, hydraulic, and stream water temperature) and biological (fish habitat) modelling at a resolution and extent important for aquatic and terrestrial organisms. We applied this approach to quantify the impacts of discharges and water temperature on habitat quality and spatial/temporal habitat use patterns of bull trout, a federally listed species along the South Fork Boise River.

We coupled process-based snow melt and hydrologic models to predict water availability within the watershed. The model fed one- and two-dimensional hydrodynamic models to predict stream hydraulics and water temperature using high-resolution (meter scale) river bathymetric data. This information was then used in an aquatic habitat modelling to characterize habitat quality distribution as a function of discharges. Our results showed that the summer thermal regime of river system would alter available habitat. The high spatial resolution analysis allows modelling to predict the importance of lateral habitats, which serve as vital refugia during high-flow events for many fish species. The advances in remote sensing, numerical modelling, and understanding of physical-biological processes provide us an opportunity to conceptualize new process-based integrated modelling tools to analyse human impacts at a catchment scale, for example, dam operation and climatic variability on aquatic habitat and status, and further to develop restoration protocols in a virtual domain before field studies are developed and/or structures built.

KEYWORDS

aquatic habitat, integrated modeling, habitat shift, lateral-channel habitat, hydraulic model, hydrologic model, water temperature

## 1 | INTRODUCTION

Water management through river-damming and reservoir operations has economic benefits and reduces flood hazards; they are two of the main human influences on freshwater ecosystems worldwide (Poff et al., 1997; Pringle, 2003). Reservoir operations affect hydrological processes by potentially altering multiple facets of the downstream riverine ecosystem including stream water flow, nutrient concentrations, water chemistry, water temperature, and sediment transport; the influence of these changes may result in beneficial conditions but often result in a decline of habitat quality and biodiversity. Changes in flow regime, whose hydrograph also depends on watershed hydrology, affect channel connectivity to floodplain, side-channel, and backwater habitats that are important to a variety of terrestrial and aquatic species. Floodplains include off-channel habitats that contain sloughs, wetlands, side channels, and other seasonally-flooded lands, which are productive for juvenile fishes (Martens & Connolly, 2014; Schlosser, 1991) and can serve as refugia, during high flows in the main channel (Bell, Duffy, & Roelofs, 2001).

In addition to anthropogenic influences, warming of climatic conditions is projected to accelerate (Intergovernmental Panel on Climate Change, 2013) and includes an increased potential for extreme drought and flood events. Climatic variability may impact the thermal regime of river systems and shift aquatic habitat distribution for a variety of species (see, Battin et al., 2007; Perry, Low, Ellis, & Reynolds, 2005). Researchers have already found increased precipitation variability and decreased snowpack that have been linked to warming, altered stream hydrology, and intensified channel disturbance result of post-fire landslides and debris flows in the North West of the United States (Isaak et al., 2010). These impacts may force changes to water resources development, operation, and management in the future. Interaction between flow releases, timing, and dam operation on stream thermal regimes are key elements for aquatic habitat quality (Carey & Zimmerman, 2014; Danehy, Colson, Parrett, & Duke, 2005; Hébert, Caissie, Satish, & El-Jabi, 2015; Johnson, Saito, Anderson, Andre, & Fontane, 2004; Lessard & Hayes, 2003; Loinaz, Davidsen, Butts, & Bauer-Gottwein, 2013; Torgersen, Price, Li, & McIntosh, 1999). The interaction between key habitat elements is often overlooked and a major gap in current environmental flow assessments (Olden & Naiman, 2010).

Large-scale climatic cycles (unregulated) and water management (regulated) shape the interaction between hydrologic, thermal, and water quality regimes in streams and rivers. These effects then cascade on habitat quality that can be quantified with in-stream flow incremental methodology (Bovee, 1982), at micro-scale or mesoscale (Costa, Martinez-Capel, Muñoz-Mas, Alcaraz-Hernández, & Gómez, 2012; Kammel, Pasternack, Massa, & Bratovich, 2016), carrying capacity (Cramer & Ackerman, 2009), or bioenergetic methods (Chipps & Wahl, 2008). Despite the need to have tools for simulating different scenarios to quantify the complex interaction among biotic and abiotic systems, based on our knowledge, only a few studies integrate large-scale processes (hydrological modelling) with fine scale processes such as aquatic meso- and micro-habitat those that do analyse impacts of human influence or climate variability on a riverine ecosystem (e.g., Kail et al., 2015; Loinaz et al., 2013; Merenlender & Matella, 2013). Merenlender and Matella (2013) integrated modelling of stream flow and aquatic habitat but did not account for watershed hydrological and water quality modelling. Loinaz et al. (2013) simulated watershed hydrology, stream water temperature, and interaction between surface and ground water but did not link them to habitat quality. Recently, Kail et al. (2015) simulated watershed hydrology and water quality using Soil and Water Assessment Tool (SWAT), river hydraulics, and change in channel morphology by 2D hydrodynamic model, whose outputs fed a spatially distributed habitat model in order to predict presence or absence of aquatic species. The integrated model that we developed in this study is what we defined an "ecohydraulics virtual watershed" because it captures large-scale physical processes and quantifies their effects on local biotic processes.

Integrated physical and ecological models (ecohydraulics) can be used to support environmental decision making because of their ability to integrate human or natural processes and environment issues in spatial as well as temporal scales (e.g., Oxley, McIntosh, Winder, Mulligan, & Engelen, 2004). They are powerful tools to examine interaction between physical properties (e.g., hydrologic, hydraulic, and geomorphologic) and ecosystems and investigate the impacts of watershed processes such as land use, water quality, quantity, and sediment dynamics on riverine habitat or ecosystem at a catchment scale (Bovee, 1982; Lancaster & Downes, 2010; Newson & Newson, 2000; Pringle, 2003; Tetzlaff et al., 2007; Zalewski, 2002). Previous research concluded that integrated modelling is the best management practice to evaluate riverine ecosystems. However, application of ecohydraulics virtual watersheds modelling has been a challenge because of data limitations, lack of understanding the relationship between physical and ecologic responses, and application at the catchment scale (McKinney, Cai, Rosegrant, Ringler, & Scott, 1999; Nilsson et al., 2003; Tetzlaff et al., 2007).

In this paper, we show that advances in remote sensing technology for mapping terrestrial landscape (LiDAR) and river bathymetric (topobathymetric LiDAR, the Experimental Airborne Advance Research Lidar), geographic information system (GIS) for visualization and data management, numerical modelling of watershed (hydrologic model), river hydraulics (hydraulic modelling), water quality (stream temperature), and biological (aquatic species) habitat models makes it possible to analyse the impacts of human and/or natural processes on ecosystems at a scale (meter scale) and extent (over several thousands of channel widths) important for aquatic organisms. These advances allowed us to develop an ecohydraulics virtual watershed, which we applied to the regulated section of South Fork Boise River (SBFR; Idaho, USA) to assess the role of discharges on rearing habitat quality and spatial distribution of the federally protected bull trout, a native cold water fish species. Our study focused on "critical rearing habitat" and in the context of this paper critical habitat is defined as foraging, migration and overwintering habitat for adult and subadult bull trout as defined under the Endangered Species Act.

## 2 | METHODS

We developed an integrated modelling framework, which links physical and biological processes to analyse ecosystem processes of river system (Figure 1). Physical model includes hydrologic, hydraulic, and water quality (temperature) modelling, whereas fish habitat represents the ecological modelling. Watershed hydrological model simulated hydrology were used (input) in hydraulic models to simulate flow hydraulics and water temperature. The predicted local flow hydraulics and water quality values were used as inputs for ecological (fish habitat) model to predict spatial and temporal aquatic habitat quality distribution (Bovee, 1982).

### 2.1 | Study area

The SFBR system (drainage area of 3,382 km$^2$) is located in South Central Idaho, Idaho, USA (Figure 2). Average width and slope of

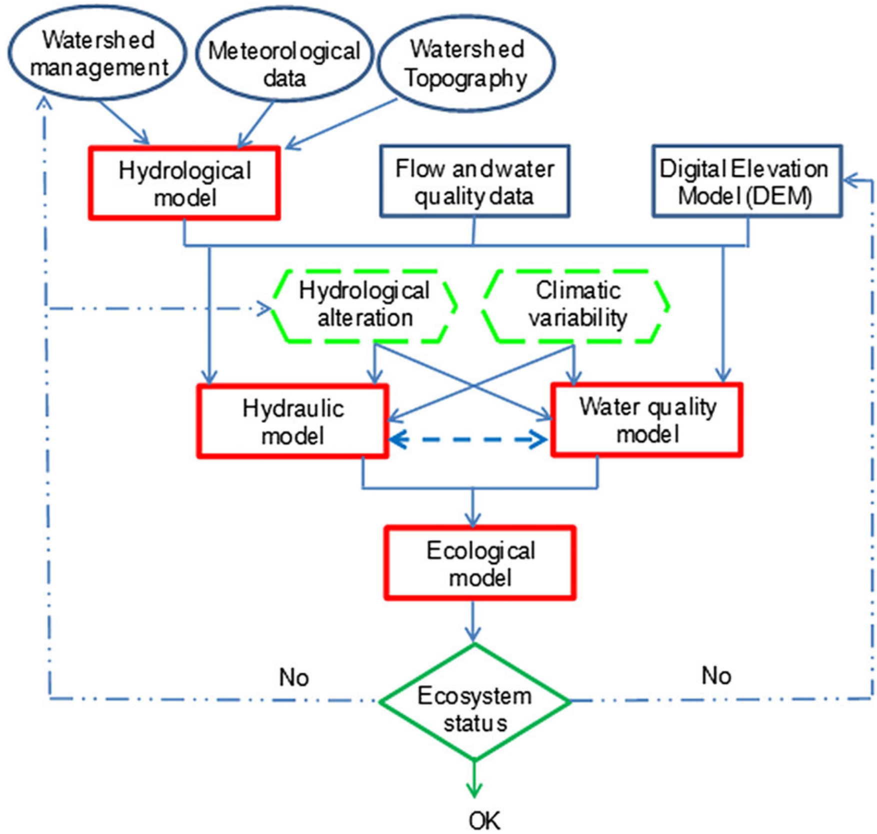


**FIGURE 1** Integrated modeling framework that couples physical and biological models to analyze status of riverine ecosystem. There is feedback between hydraulic and water quality model

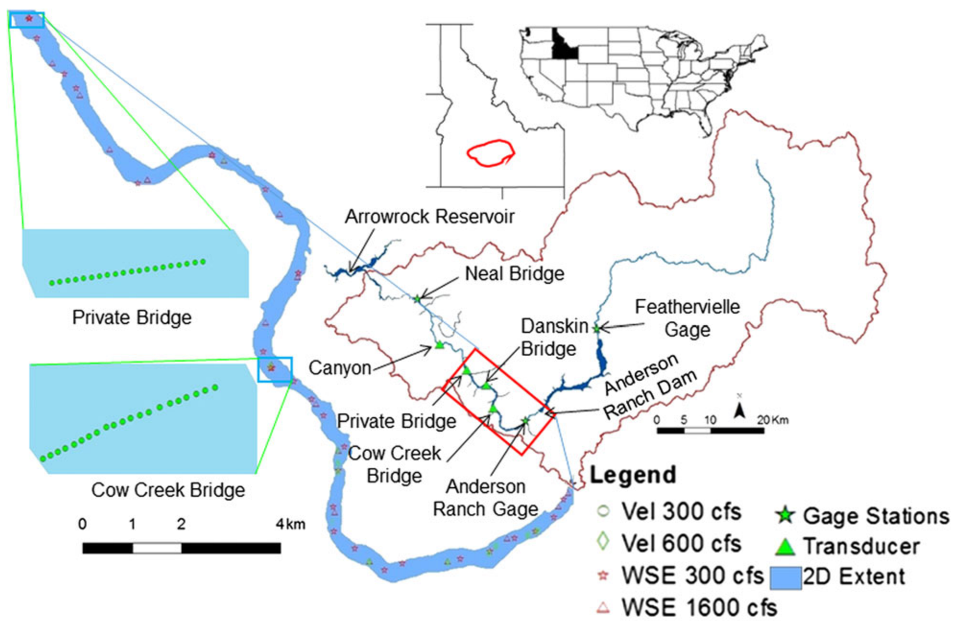


**FIGURE 2** Study area, transducer, gage station, water surface elevation and velocity measurement locations. Habitat model area is shown with rectangular box

the river is approximately 41 m and 0.0043, respectively. It hosts habitat that is federally protected and considered critical for a migratory population of bull trout. The Anderson Ranch Dam regulates downstream flows and is operated for irrigation, flood control, and power production. The SFBR below the dam can be divided into a 23-km upper reach (study site) that starts from the Anderson Ranch Dam and a 22-km lower reach that ends at the Arrowrock Reservoir. The study site is dominated by side channels, braided channels, deep pools, riffles, and runs. The main channel connects with side channels, which are mostly ephemeral during high flows, and may work as backwater areas during low flows with a connection only in the downstream end. The lower 22-km reach flows almost entirely through a canyon and exhibits different habitat features than the upper reach.

### 2.2 | Hydrological model

We coupled an energy balance snow model (ISNOBAL; Marks, Domingo, Susong, Link, & Garen, 1999), which estimates spatial distribution of snow accumulation with a hydrologic model (Penn State Integrated Hydrology Model [PIHM]) to predict flows for ungaged streams and tributaries (Kumar, Duffy, & Salvage, 2009; Yu, Bhatt, Duffy, & Shi, 2013). The ISNOBAL model accounts for evapotranspiration, sublimation, condensation, and snow drifting based on air temperature, wind speed and direction, precipitation, solar radiation, and relative humidity to differentiate precipitations into snow or rain (Sohrabi, 2016). PIHM is a fully coupled and semi-distributed hydrologic model and simulates hydrological processes (Kumar et al., 2009; Yu et al., 2013).

The coupled PIHM and ISNOBAL models were run with a high-resolution grid of 50 × 50 m cells over the South Fork River Basin. The PIHM was calibrated (water year 2010) and validated (for 2006, 2007, and 2013 water years) at Featherville (U.S. Geological Survey gage 13186000) and at the Anderson Ranch Dam (U.S. Geological Survey gage 13190500) for 2006, 2007, 2010, and 2013 water years.

The PIHM predicted hydrological patterns were similar with observed hydrological patterns at both Feathervielle and Anderson Ranch Dam gages. Both stations exhibit Nash–Sutcliffe coefficient higher than 0.74 for all water years similar to other studies (Kumar, Marks, Dozier, Reba, & Winstral, 2013; Moriasi et al., 2007; Yu et al., 2013) except in 2006 water year when the percent bias (PBIAS) were around 4% for model calibration and validation. The model underestimated the highest peak for 2006, which resulted in an error of 23% and 21% at the Anderson Ranch Dam and Featherville gage stations, respectively (Figure 3).

### 2.3 | Hydraulic model

We used the Danish Hydraulic Institute (DHI) software MIKE 11 (1D) for developing water temperature model to simulate stream water temperature along the SFBR, which uses an implicit, finite-difference algorithm to solve the unsteady one-dimensional de Saint-Venant equations (DHI, 2011a). The software (MIKE11) dynamically couples a one-dimensional advection-dispersion heat transport equation and 1D hydraulic models to predict water temperature distribution along stream. The high-resolution cross-sections for 1D model were extracted at every 30 m from the 1 m resolution digital elevation model. The temperature model considers solar radiation, evaporation, convection, and the atmospheric heat balance (DHI, 2011a). We used MIKE21 (DHI, 2011b) software to develop a 2D hydraulic model to simulate flow velocities (depth averaged), depths, and water surface elevations (WSEs). A digital elevation model (2 × 2 m resolution) of floodplain and channel (submerged) bathymetry, surveyed with the Experimental Airborne Advance Research Lidar (McKean et al., 2009), was used to develop the 2D model.

#### 2.3.1 | One-dimensional hydraulic and temperature model

The one-dimensional hydraulic and temperature model represents a river reach (45 km) between the Anderson Ranch gage station (United States Geological Survey [USGS] 13190500) and Neal Bridge gage station (USGS 13192200; Figure 2). The hydrologic (Anderson Ranch and Neal Bridge USGS Gage stations) and water temperature (U.S. Bureau of Reclamation gage station at the Anderson Ranch Dam) data were used as upstream (discharge and temperature) and downstream (stage and temperature) boundary conditions, respectively. We used lateral inputs only from tributaries with watershed area higher than 9 $km^2$ (Figure 2). Measured discharge and temperature time series are available for some of these tributaries for the period after 2011. Discharges and stream water temperatures for ungaged tributaries were estimated with the hydrological model (Sohrabi, 2016) and the Stream Water Temperature Model (Sohrabi, Benjankar, Tonina, Wenger, & Isaak, 2017) models, respectively.

We compared model simulated and field measured WSE for model calibration for discharges 8.5 and 45.6 $m^3\ S^{-1}$, which resulted in spatially varied (three zones) Manning's roughness *n* values (0.035–0.15) within a cross-section similar to other studies (Acrement & Schneider, 1989; Benjankar, Tonina, & McKean, 2014; Horritt & Bates, 2002; Zink & Jennings, 2014). Root mean square error (RMSE) were 0.12 m at random locations for both discharges (Figure 4 and Table 1). For model validation, we compared water wave travel at three locations: Cow Creek Bridge, Private Bridge,

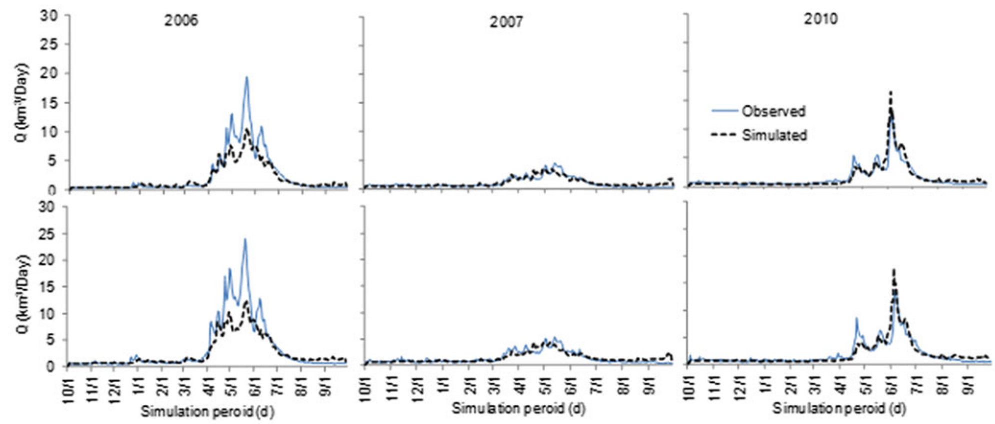


**FIGURE 3** Simulated and observed hydrographs for the Featherville (top) and the Anderson Ranch gage stations

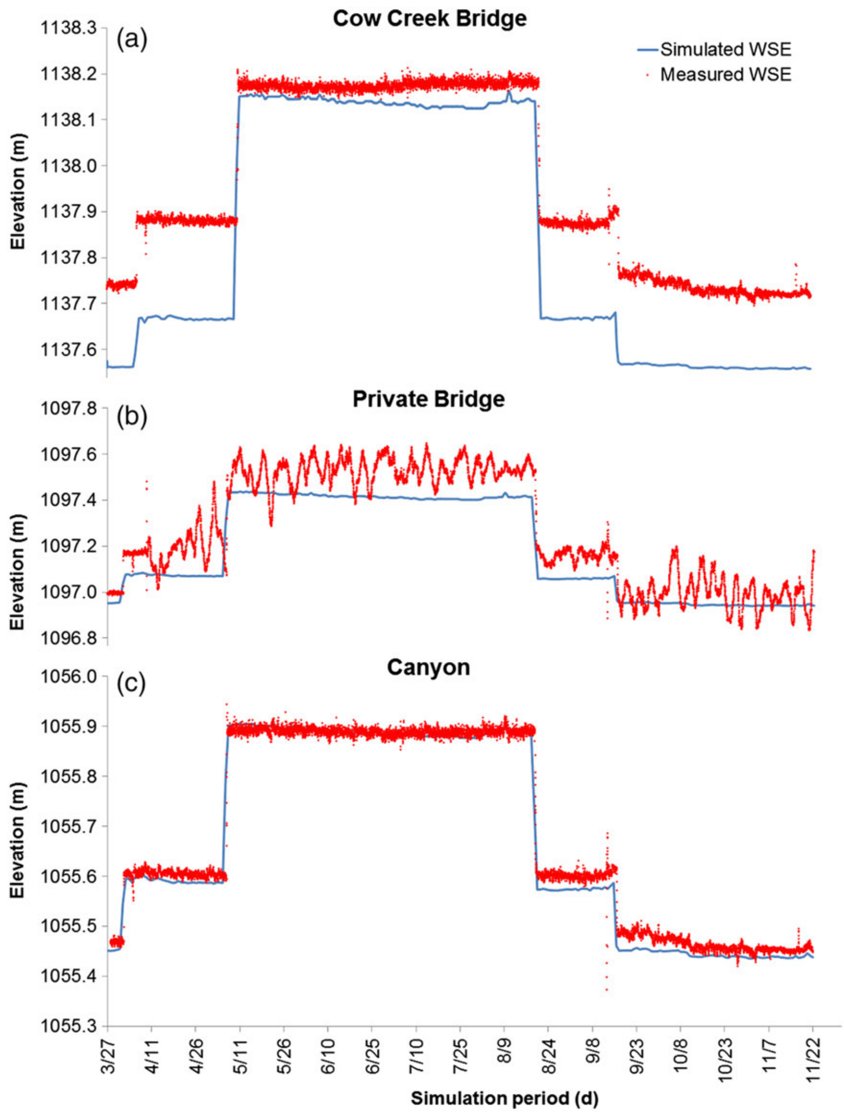


**FIGURE 4** Measured and simulated WSE for 30-min time series (discharge varies from 8.5 to 46 $m^3 S^{-1}$) at (a) the Cow Creek Bridge, (b) Private Bridge, and (c) Canyon section

and Canyon including measured and simulated WSEs (see Figure 2 for their location along the reach) for the period between 3/22/2013 and 11/22/2013 (Figure 4). Comparisons of daily WSEs estimated RMSE of 0.15, 0.11, and 0.02 m at the Cow Creek Bridge, Private Bridge, and Canyon locations, respectively, whereas RMSE were 0.15, 0.12, and 0.02 m for 30-min time steps (Table 1). Simulated WSE patterns matched with the measured patterns at all three locations (Figure 4a–c) including the flow-wave patterns. Our analyses of topographical data showed that LiDAR surveyed elevations were 0.24 m lower (after bias correction) than real-time kinematic Global Positioning System (RTK GPS) surveyed elevations near the Cow Creek Bridge, which could explain the systematic lower simulated than observed WSE at the Cow Creek Bridge.

We used 30 min temperature time series measured at a short distance below the Anderson Ranch Dam as the upstream boundary for the temperature model. Hourly resolution meteorological data were used to simulate heat exchanges between atmosphere and stream. We used relative humidity (hourly), air temperatures (15 min), and solar radiation (hourly) from Boise airport, Arrowrock Dam, and Bearskin Creek weather stations, respectively. Model parameters including calculated daily sunlight hour were modified to match the diel temperature fluctuations observed in the river.

RMSEs were 0.72, 0.70, 0.85, and 0.91°C for daily average water temperatures and 1.23, 1.33, 1.58, and 1.43 for 30-min temporal resolution at the Cow Creek Bridge, Danskin Bridge, Private Bridge, and Canyon locations, respectively (Figure 5 and Table 1). The model simulated results (RMSE < 1.58°C) were consistent with other studies (Hébert et al., 2015; Loinaz et al., 2013; Wang, Yang, & Khangaonkar, 2010).

### 2.3.2 | Two-dimensional model

A 2 m grid size 2D hydraulic model was developed for the study site to simulate flow hydraulics (Figure 2). We adjusted Manning's roughness during model calibration by comparing observed and predicted WSEs and velocities for 8.5 $m^3 S^{-1}$ flow. Velocity measurements were taken at multiple random fast and slow river velocity locations along the channel and cross section (Figure 2). We validated the model for discharges of 17 and 46 $m^3 S^{-1}$ comparing simulated and measured WSEs, whereas only 17 $m^3 S^{-1}$ for velocity. We were not able to

**TABLE 1** Errors associated 1D and 2D hydraulic model calibration and validation

| Locations | | 1D hydraulic | | 1D temperature | | |
|---|---|---|---|---|---|---|
| | | Daily | 30 mins | Daily | 30 mins | 2D HD |
| [f]Random Locations | WSE (m) | 0.12[b]<br>0.12[d] | | | | 0.2[b]<br>0.18[c]<br>0.17[d] |
| | Vel (m/s) | | | | | 0.27[c] |
| [g]Cow Creek | WSE | 0.15 | 0.15 | | | |
| | Temp ($^0$C) | | | 0.72 (0.98[e]) | 1.23 (0.90[e]) | |
| | [a]Vel (m/s) | | | | | 0.07[b] |
| [g]Danskin Bridge | WSE (m) | | | | | |
| | Temp ($^0$C) | | | 0.70 (0.96[e]) | 1.33 (0.87[e]) | |
| [g]Private Bridge | WSE (m) | 0.11 | 0.12 | | | |
| | Temp ($^0$C) | | | 0.85 (0.93[e]) | 1.58 (0.81[e]) | |
| | [a]Vel (m/s) | | | | | 0.10[b] |
| [g]Canyon | WSE (m) | 0.02 | 0.02 | | | |
| | Temp ($^0$C) | | | 0.91 (0.92[e]) | 1.43 (0.84[e]) | |

*Note*. WSE: water surface elevation.

[a]Velocity measured at cross-section.

[b]Discharge 8.5 $m^3 S^{-1}$.

[c]Discharge 17 $m^3 S^{-1}$.

[d]Discharge 46 $m^3 S^{-1}$.

[e]Coefficient of determination ($R^2$).

[f]Water surface elevation and velocity measured at random locations.

[g]Measured water surface elevation and temperature time series.

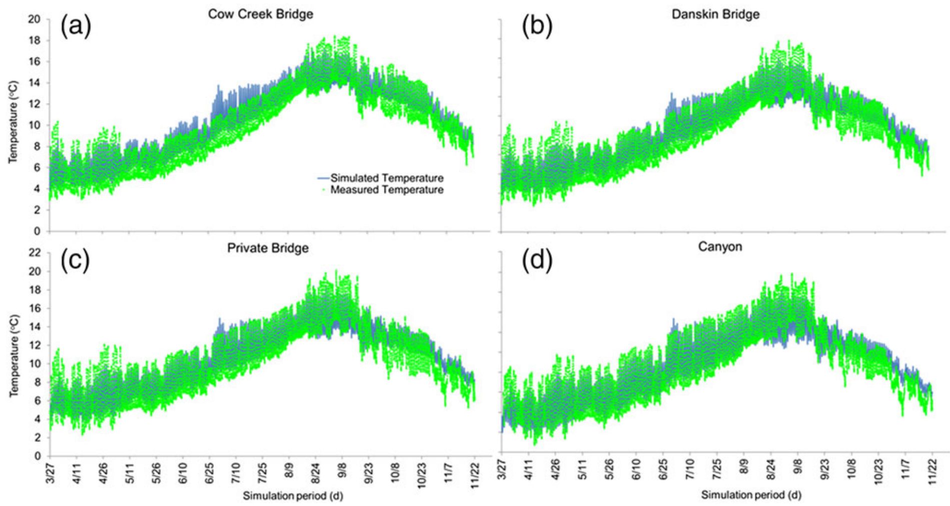


**FIGURE 5** Measured and simulated 30 min interval water temperatures for the (a) Cow Creek Bridge, (b) Danskin Bridge, (c) Private Bridge, and (d) Canyon locations

measure velocity for flows higher than 17 $m^3 S^{-1}$ because of safety concerns.

WSE RMSEs at random locations were 0.2, 0.18, and 0.17 m for flows of 8.5, 17, and 46 $m^3 S^{-1}$, respectively (Table 1). The RMSEs were 0.27 m $s^{-1}$ for velocity at random locations for discharge 17 $m^3 S^{-1}$ (Table 1). Flow velocity RMSEs ($R^2$ = 0.77) were 0.07 and 0.1 m $s^{-1}$, at the Cow Creek Bridge and Private Bridge, respectively, for discharge 8.5 $m^3 S^{-1}$ (Figure 6a,b and Table 1). Furthermore, the model was able to replicate flow-wave patterns similar to the observed (Figure 6c). The model performance to simulate both WSE and velocities for different flows were consistent with other studies (e.g., Boavida, Santos, Katopodis, Ferreira, & Pinheiro, 2013; Guay et al., 2000; Pasternack, Wang, & Merz, 2004; Tarbet & Hardy, 1996). The model overestimated the measured velocities at 17 $m^3 S^{-1}$ at random location (Figure 6d). However, at this high flow, velocities were too fast and with high turbulence; in these conditions, it is challenging to minimize reading error for manually operated equipment. Thus, measurements might be affected by high uncertainty because of the difficulty of working in that environment (Nystrom, Rehmann, & Oberg, 2007; Waddle, 2010). Additionally, measured flow velocities are influenced by localized 3D morphology and highly variable substrate size distribution not captured at the scale of the cell and predicted velocities are affected by interpolation technique (see Guay, Boisclair, Leclerc,

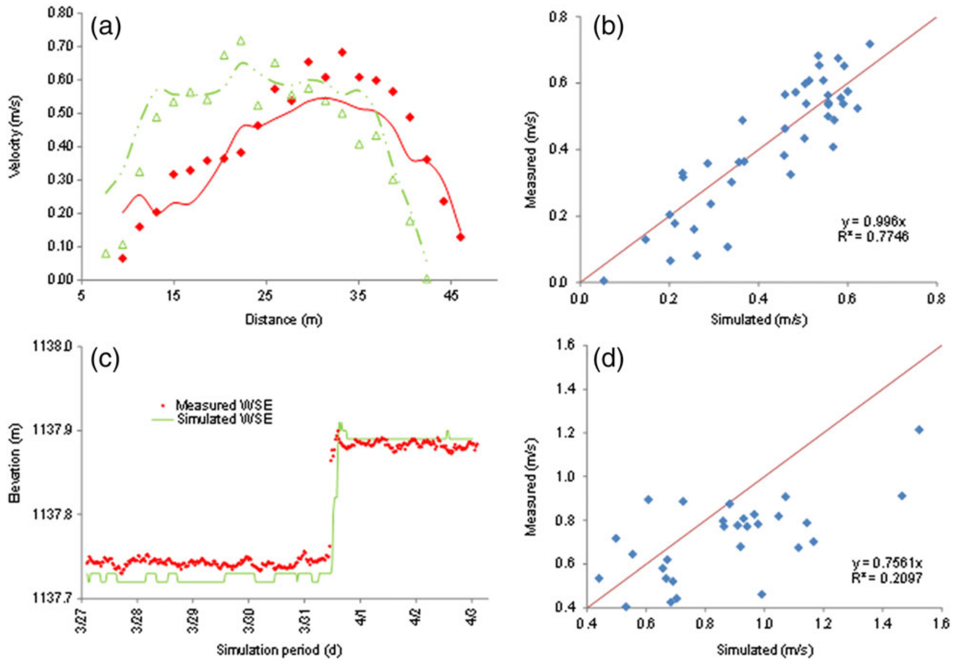


**FIGURE 6** 2D model simulated and measured and velocities for 8.5 $m^3$ $S^{-1}$ at the (a) Cow Creek and Private Bridges, (b) 1:1 diagram for measured and simulated velocity for the Cow Creek and Private Bridges, (c) wave-travel patterns for measured and simulated water surface elevations (WSEs) at the Cow Creek Bridge, and (d) 1:1 diagram for measured and simulated velocity at random locations along the reach for 17 $m^3$ $S^{-1}$

Lapointe, & Legendre, 2001; Papanicolaou, Elhakeem, & Wardman, 2011; Pasternack, Wang, & Merz, 2003).

## 2.4 | Biological model

### 2.4.1 | Preference curves

Bull trout is a cold water obligate species and sensitive to temperature change (Fraley & Shepard, 1989; Rieman & McIntyre, 1993); thermal requirements may vary with geographic area and life-history expression (Anglin et al., 2004; Hillman & Essig, 1998; Howell, Dunham, & Sankovich, 2010; Poole et al., 2001; Selong, McMahon, Zale, & Barrows, 2001). Furthermore, it can also be influenced by behavioural cues and life history; developmental and migratory stage may have different thermal tolerances. Although ideal bull trout survival and reproduction temperature is less than 15°C, Hillman and Essig (1998) and Howell et al. (2010) have shown that they can adapt their life cycle to a wide range of temperature thresholds. The Environmental Protection Agency recommends a temperature range between 12 and 14°C, which differs from the 4 to 10°C optimal temperature for juvenile bull trout growth and rearing habitat. This higher range seems to be supported by the laboratory experiment conducted by Selong et al. (2001) and Mesa, Weiland, Christiansen, Sauter, and Beauchamp (2013), who observed assimilation and optimal growth rate between 8 and 20°C and between 10.9 and 15.4°C, respectively. U.S. Forest and Wildlife Services defined the physical or biological features essential for conservation ranging from 2 to 15°C.

Previous studies concluded that daily maximum temperature (DMT) above a critical threshold may be the major indicator for distribution of aquatic species (Olden & Naiman, 2010). Therefore, we used DMT as a metric for temperature suitability index ($SI_t$) for our study (Abatzoglou, Barbero, Wolf, & Holden, 2014). We identified $SI_t$ for water temperatures <16 and >22°C are 1 and 0, respectively, whereas its value linearly decreases from 1 to 0 between 16 and 22°C, (Figure 7, right) (Bonneau & Scarnecchia, 1996; Maret & Schultz, 2013; McMahon, Zale, Barrows, & Selong, 2007; Poole et al., 2001; Selong et al., 2001). DMTs do not change substantially spatially along the SFBR within a day (Benjankar et al., 2018); therefore, we use a single $SI_t$ for this study (see Maret, Hortness, & Ott, 2006). This approach might introduce some uncertainties when DMTs changes between 16 to 22°C along the streams, which was not observed for our study.

We adopted rearing habitat preference curves for depth and velocity from previous studies (Lewis River workshops, 2000; Washington Department of Fish and Wildlife, 2004). Although, these preference curves are developed for streams of Washington State, observed (at bull trout locations) water depths and velocities fitted well within the adopted curves (Figure 7, left). U.S. Bureau of Reclamation surveyed bull trout locations by using radio telemetry.

### 2.4.2 | Model development

We developed a biological (fish habitat) model in a GIS framework to quantify habitat quality using simulated water depth, velocity, temperature, and univariate rearing habitat preference curve (Benjankar et al., 2018). The model quantifies spatially distributed habitat suitability index (HSI), an indicator of habitat quality at the cell scale (2-m grid size), and weighted useable area (WUA), an indicator for quality at the river segment scale ($10^2$ channel width; Bovee, 1978). We specifically selected rearing habitat because the study reach is known as a rearing habitat for migratory bull trout year around. We calculated SI

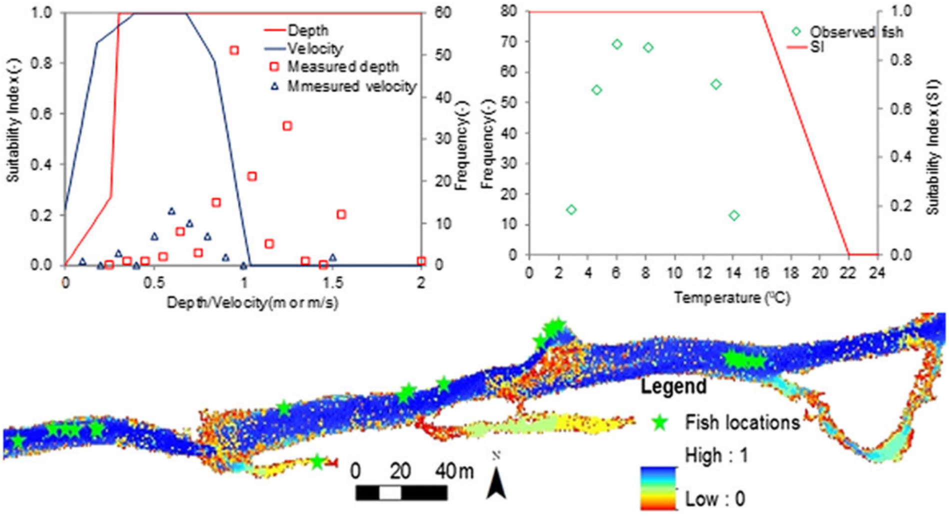


**FIGURE 7** Water depth and velocity preference curves for bull trout adult, red squares, and blue triangles are measured depths and velocities (frequency) at fish locations (left). Water temperature preference curve and green diamonds are water temperature (frequency) at fish locations (right). Observed fish distribution over simulated habitat quality

for each grid cell and WUA for each day based on measured daily average flow at the Anderson Ranch Dam gage station as described in the study of Benjankar et al. (2018). We used GIS-based fish observed locations to validate fish habitat model. Majority (84%) of fish locations were overlapped with the high-quality (0.8–1) simulated habitat for flow 8 $m^3 S^{-1}$. (Figure 7).

## 2.5 | Discharge and habitat

We simulated distributed water depth and velocity with the 2D hydraulic model for the following discharges 4, 8, 17, 28, 45, 57, 68, 85, 102, 142, 184, and 227 $m^3 S^{-1}$ and then calculated area-weighted median depth and velocity to analyse relationship between discharges and hydraulics (Benjankar et al., 2018). These discharges cover the entire flow regime during both regulated (dam releases) and unregulated (without dam and reservoir) flows. The discharge of 8 $m^3 S^{-1}$ is the present minimum flow release through the Anderson Ranch Dam, whereas 227 $m^3 S^{-1}$ (2006) is the maximum discharge measured at the USGS Featherville gage station (natural) during the period between 1946 and 2012. Furthermore, we considered a hypothetical minimum release of 4 $m^3 S^{-1}$ to study the potential impact of dam management during a drought event on aquatic habitat. We used a GIS-based fish habitat model to simulate spatially distributed HSI and change in WUA (Q-WUA curve) based on hydraulics for all discharges.

## 2.6 | Habitat prediction

We developed habitat time series of WUA for an entire water year (i.e., 2013) based on daily stream flows at the Anderson Ranch gage station, water depth, and velocities in each grid cell and DMT (Benjankar et al., 2018). Habitat time series has been used in other studies as an indicator for temporal habitat status (Li, Chen, Tonina, & Cai, 2015; Muhlfeld et al., 2012). We compared habitat (WUA) and hydraulic-based habitat (WUA) time series to analyse the effect of temperature on habitat prediction. Habitat (WUA) time series considers water depth, velocity, and stream temperature, whereas hydraulic-based habitat considers hydraulic variables water depth and velocity alone. Furthermore, we compared habitat and hydraulic-based habitat time series for regulated and unregulated scenarios, which indicate temperature and flow regimes with and without the Anderson Ranch Dam operation impacts. We assigned water temperature $SI_t$ of 1 to estimate hydraulic-based habitat.

For habitat distribution, we separated the simulated spatially distributed HSI and WUA between in-channel habitat, that is, within the bankfull channel (here after in-channel) and lateral habitat, that is, beyond the bankfull wetted area extension (here after lateral channel). The later includes side channels, back water areas and inundated floodplain. We delineated the spatial extent of bankfull channel using the River Bathymetry Toolkit (McKean et al., 2009). We defined low and high flows those below and above bankfull flow (68 $m^3 S^{-1}$), respectively.

# 3 | RESULTS

## 3.1 | Hydraulics

Inundated areas increased slowly up to the bankfull discharge and then above bankfull rapidly expanded into lateral-channels and on floodplains (Figure 8). The inundation area for the maximum flow (227 $m^3 S^{-1}$) was twice (2.18 $km^2$) that of bankfull discharge (1.15 $km^2$). The median (50 percentile) water depth increased with high flows, except for 227 $m^3 S^{-1}$ (Figure 8). Furthermore, ranges of minimum (5 percentile) and maximum (95 percentile) depths increased with high discharges.

As with water depths, range of minimum and maximum velocities increased with higher discharges (Figure 8). Fast flow velocities (larger than 1 $m s^{-1}$) simulated in in-channel zones, whereas slow velocities (lower than 1 $m s^{-1}$) in lateral-channel zones (Figures 8 and 9) for discharges higher than bankfull. Area-weighted median velocity

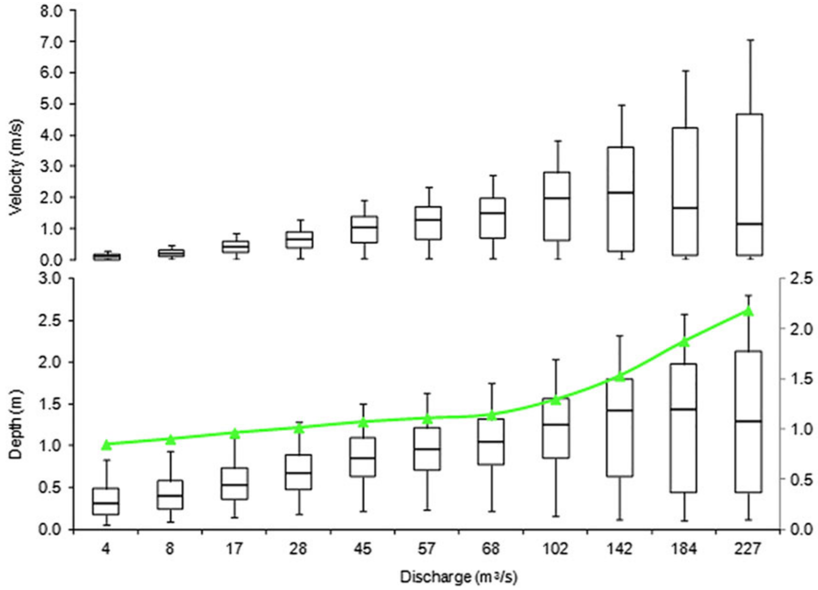


**FIGURE 8** Box plots for minimum (5 percentile), first quartile (25 percentile), median (50 percentile), third quartile (75 percentile), and maximum (95 percentile) of water depths (bottom panel) and velocities (top panel) for different discharges. Inundated areas as function of discharge (green line)

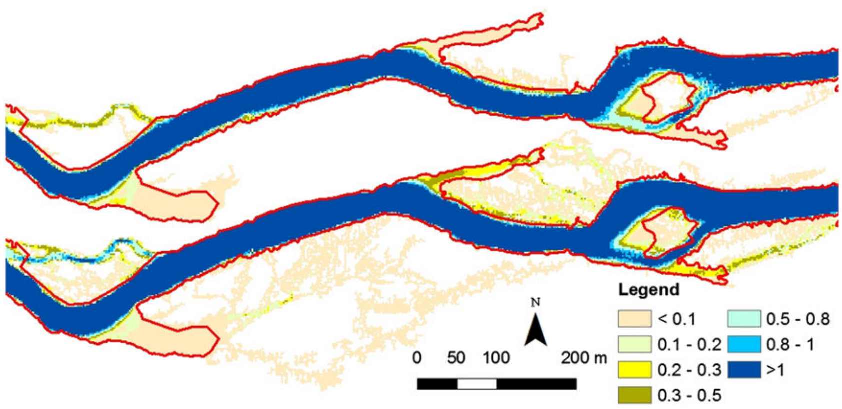


**FIGURE 9** Spatial distribution of velocities (m $S^{-1}$) and inundated areas for discharges 68 $m^3$ $S^{-1}$ (top panel) and 102 $m^3$ $S^{-1}$ (bottom panel). Red lines indicates bankfull channel boundary

increased with high flows, except for flows 184 and 227 $m^3$ $S^{-1}$, where it decreased. This indicated the in-channel flow connected with side-channels, where velocities are slower than in the in-channel zone, and increased inundation areas extensively for large flows (Figure 9).

## 3.2 | Habitat

### 3.2.1 | Available habitat

The quality of habitat varied spatially with higher discharges (Figure 10, right). The connectivity between in-channel with lateral habitats (side channels and floodplain) during high flows substantially improved bull trout rearing habitat (Figures 8–10). Optimal rearing bull trout habitats peaked at the discharge of 17 $m^3$ $S^{-1}$ based on both WUA and HSI, whose values then decreased for discharges less than 17 $m^3$ $S^{-1}$ primarily due to reduction in flow depths (Figures 8–10) and for discharges higher than 17 $m^3$ $S^{-1}$ up to the bankfull elevation, above which WUA and HSI increased slowly again (Figure 10, left). The increase in WUA was considerable compare to HSI (0.21–0.33), due to an increase in wetted areas (Figure 8–10) rather than habitat quality (HSI) at high flows.

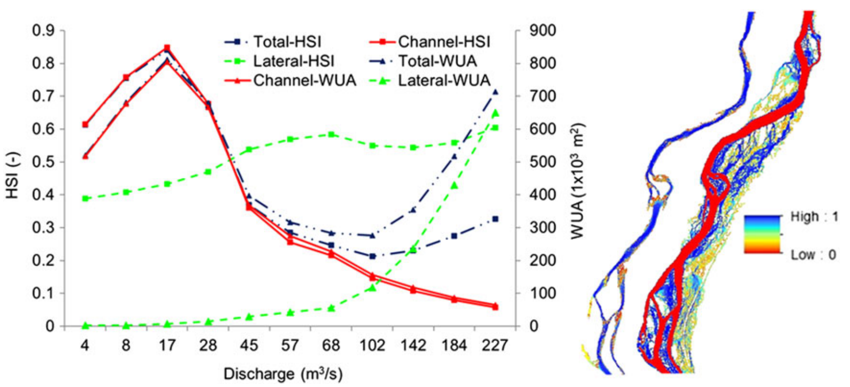


**FIGURE 10** Weighted useable area (WUA) and habitat suitability index (HSI) quantified with water depth and velocity only (without temperature) as a function of discharges calculated for channel, off-channel, and combined spatial extents (left) and spatial shift in high quality habitat for low (8 $m^3$ $S^{-1}$) and high flows (102 $m^3$ $S^{-1}$; right)

### 3.2.2 | Temperature effect

During July, August, and September for the unregulated scenario, habitat qualities (with temperature) were noticeably lower than the hydraulic-based habitat (without temperature) qualities due to stream water daily maximum water temperature being larger than the threshold value for optimal temperature (16°C; Figure 11, left). Stream water DMTs were less than 16°C in other months (October–June), resulting with $SI_t$ = 1, and consequently both habitat and hydraulic-based habitat qualities were similar. Conversely, for the regulated scenario stream water DMTs were less than 16°C (Figure 11, right) regardless of the period of the year for. Therefore, both habitat and hydraulic-based habitat qualities were always similar (Figure 11, left).

### 3.2.3 | Spatial habitat shift

The spatial distribution of rearing habitat quality was quite homogenous and mostly of high quality in the in-channel zone at low discharges (less than bankfull; Figure 10, right). However, at high flows (above 17 $m^3\ S^{-1}$), good quality habitat shifted towards lateral channel (on floodplain and in side channels) as discharge rises. In-channel HSI and WUA values increased to a peak for 17 $m^3\ S^{-1}$ and then only decreased (Figure 10, left). Conversely, lateral-channel WUA values were low at low flows and increased almost exponentially for high discharges. Nevertheless, HSI values increased slowly (0.39–0.61) but monotonically.

## 4 | DISCUSSIONS

### 4.1 | Integrated Modeling

Previous research shows that integrated modelling is a powerful approach to analyse riverine ecosystems for biotic and abiotic alteration (see Merenlender & Matella, 2013), but such integrated modelling approaches face challenges of data limitations, model connectivity, and interrelationship between the physical and biological processes (Boavida et al., 2013). Here, we show that advances in remote sensing to map both emerged and submerged topography linked to hydrological and hydraulic modelling, whose advances made them faster, and provide a powerful tool to analyse fish habitat at the meter scale resolution along long section of streams (e.g., 23 km). These advances made integrated physical and biological models accessible and flexible to assess impact of different land use practices on hydrology and water quality and to assess impacts of dam operation on the riverine ecosystem (e.g., Abdulla, Lettenmaier, Wood, & Smith, 1996; Benjankar et al., 2012; Benjankar, Egger, Jorde, Goodwin, & Glenn, 2011; Li et al., 2015; VanShaar, Haddeland, & Lettenmaier, 2002). Integrated models can be useful for designing ecosystem restoration projects, considering both physical, and biological as well as spatial and temporal scales. Reservoir dynamic models, such as Estuary, Lake, and Coastal Ocean Model (Vilhena, Hillmer, & Imberger, 2010) could be a potentially critical piece in integrated models as shown in other research (Marzadri, Tonina, Mckean, & Tiedemann, 2012), which can simulate reservoir outflow water quality parameters (e.g., water temperature and nutrient; Johnson et al., 2004; Romero, Antenucci, & Imberger, 2004). Integrated modelling that includes reservoir dynamic models may be extremely helpful in exploring the effect of future condition that may be influenced by management of contractual water, and climatic variability.

### 4.2 | Habitat

Application of the integrated modelling allowed us to show that bull trout rearing habitat quality (distribution and floodplain connectivity) is sensitive to changes in river discharge and temperature regime. Minimum flow release less than the current minimum flow (8 $m^3\ S^{-1}$) will reduce the available bull trout rearing habitat. Interestingly, the flow (17 $m^3\ S^{-1}$), which maximizes the in-channel habitat, is also the minimum flow release during late-spring and summer months (April–September) to fulfil the contractual irrigation water demand during average climatic years. Higher discharges may occur depending on water volume in the reservoir and climate variability.

Discharges during the summer for regulated scenarios are consistently cold water released from the lowest stratum of the reservoir; therefore, stream temperature may not be a critical parameter for bull trout rearing habitat in the SFBR downstream from the Anderson Ranch Dam. A portion of the migratory bull trout population uses the SFBR for rearing habitat throughout the year. Bull trout that over winter in the Arrowrock Reservoir leave the reservoir migrating to spawning locations in the Middle and North Forks of the Boise River or as water temperature in the reservoir increases migrate to the SFBR

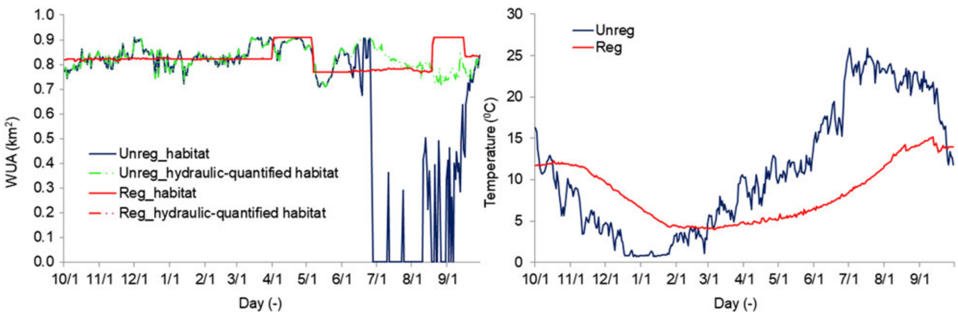


**FIGURE 11** Dam regulated and unregulated (natural) weighted usable area (WUA; left). Hydraulic-quantified habitat in legend indicates omission of water temperature for WUA calculation. Water temperature patterns for the water year 2013 for unregulated and regulated scenarios (right)

to over summer (Salow & Hostettler, 2004) before returning to the reservoir in November (U. S. Bureau of Reclamation, 2013).

Our results showed that discharge and water temperature manipulation through changing dam operation patterns (Figure 11) can maintain aquatic habitat, for instance in offsetting impacts of future climatic variability (Benjankar et al., 2018). Conversely, stream water temperatures may be a critical factor for bull trout rearing habitat in the unregulated scenario for much of the summer (July, August and September). Thus, historically as shown by the unregulated case, bull trout may not have used this portion of SFBR for summer rearing habitat due to high temperatures (>22°C), suggesting that they may had migrated upstream or to tributaries (Figure 11, right).

This result confirmed the importance of considering water temperature variable, which depends on timing of flow releases on stream thermal regime to analyze aquatic habitat quality, specifically during the warm summer months (Olden & Naiman, 2010). One of the major findings of our study is that the habitat quality analyses may over predict available habitat for a selected species based only on hydraulic variables without considering water quality (temperature) (Figure 11, left). For example, habitat quality based on water depths and velocities is relatively higher than for the habitat quantified using water depth, velocity and temperature for unregulated scenario.

The spatial distribution of quality habitat shifts from in-channel to lateral-channel areas, which underscores the importance of the lateral-channel habitat. At high flows, fish may seek hydraulic refugia from high flow velocities in the lateral-channel (Bell et al., 2001; Shirvell, 1994). Therefore, bull trout and other fish species (e.g., Sculpins, Dace, Kokanee, Mountain Whitefish and Rainbow Trout) that resident in the SFBR system could use lateral-channels during high flows (U. S. Bureau of Reclamation, 2013). Salmonids also use lateral-channel habitats to avoid predation and reduce competitions from other species (Shirvell, 1994) as well as for rearing during winter months (Solazzi, Nickelson, Johnson, & Rodgers, 2000); thus this habitat type, it serves a significant purpose seasonally and developmentally for all fishes. High flows, larger than bankfull, are important because they shape the stream morphology, clean gravel from fine sediment, mobilize and transport sediments, and maintain stream-floodplain connectivity. However, regulated discharges are typically below bankfull for most of the year. This may lead to reduced or a loss of stream-lateral habitat connectivity because of the low frequency of bankfull flows. Consequently, conservation and restoration of lateral-channel habitats is necessary and requires periodic high discharges to maintain connectivity (Bowen, Bovee, & Waddle, 2003; Gorski, Collier, Hamilton, & Hicks, 2014).

## 5 | CONCLUSIONS

Our process-based integrated modelling framework of an "ecohydraulic virtual watershed" has the potential to analyse the impacts of anthropogenic activities, for example, dam operation, restoration project or climate variability on the habitat and ecosystem of aquatic organisms. An ecohydraulic virtual watershed approach is able to predict the status or change in physical processes and the resulting effects on a riverine ecosystem as a result of cascading impacts such as watershed management, hydrological alteration, and climate variability. This approach could provide information that allows water managers to simultaneously meet their contractual and regulatory purposes and offset the impact of climate variability on aquatic ecosystems. This approach could also be advantageous to ecosystem restoration processes because it has the ability to analyse the effectiveness of different restorations projects (i.e., modification of watershed management or stream morphology and flow through man-made structures such as dam and reservoir) before they are implemented.

Our model applied to the SFBR illustrates the importance of stream water temperature for quantifying aquatic habitat, especially for water temperature sensitive species like bull trout. Contrary to expectations, our results showed that bull trout habitat might not be limited by present reservoir operations although they altered historic stream water temperature and discharge regimes to some extent. Although, in-channel habitat quality decreases with increasing discharge, higher quality habitat shifts to lateral channels as high flows reconnects lateral channel with the main channel.

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