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Modelling changes in trout habitat following stream restoration

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Abstract

Stream restoration was implemented on the Upper Arkansas River near Leadville, Colorado, to improve brown trout (*Salmo trutta*) populations. Metals pollution and channel disturbance associated with historic mining, land use, and water development degraded aquatic and riparian habitat. Changes in instream habitat quality following restoration were investigated with a before–after–control–impact study design. Baseline, as-built, and effectiveness surveys were conducted in 2013, 2014, and 2016, respectively. Two-dimensional hydrodynamic modelling with River2D was used to estimate weighted usable area (WUA) for adult, juvenile, fry, and spawning brown trout across a range of flows. WUA was calculated from habitat suitability curves for velocity, depth, and channel substrate. Foraging positions (FP) and habitat heterogeneity were also evaluated as indices of habitat quality. All results were analysed with analysis of variance. At impact sites, WUA increased by 12.2% from 2013 to 2014 but decreased by 10.2% from 2014 to 2016, whereas FP increased by 24.8% from 2013 to 2014 but decreased by 26.1% from 2014 to 2016. Spawning habitat increased 53.3% from 2014 to 2016 at impact sites. The 15.4% increase in depth variability from 2013 to 2016 indicates that habitat heterogeneity was enhanced at impact sites. No changes in WUA, FP, or habitat heterogeneity were observed at control sites. Although changes in WUA and FP suggest that initial habitat improvements were not sustained, increased spawning habitat and depth heterogeneity suggest otherwise. Our results highlight the value of monitoring strategies that utilize multiple lines of evidence to evaluate restoration effectiveness, inform adaptive management, and improve restoration practices.

KEYWORDS

aquatic habitat, brown trout, foraging positions, hydrodynamic modelling, River2D, stream restoration, weighted usable area

1 | INTRODUCTION

Although improved fish habitat and populations have been documented following stream restoration (Pierce, Podner, & Carim, 2013; Schmetterling & Pierce, 1999; Whiteway, Biron, Zimmermann, Venter, & Grant, 2010), monitoring of projects is rare (Bash & Ryan, 2002; Bennett et al., 2016; Bernhardt et al., 2007; Roni et al., 2002; Whiteway et al., 2010), and the long-term benefits remain uncertain. Studies have documented negative impacts associated with unnecessary (Kauffman, Beschta, Otting, & Lytjen, 1997), unmaintained, or damaged restoration projects (Thompson, 2002). Other studies

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report disparate and variable success rates for common restoration treatments (Miller & Kochel, 2012, 2009; Roni et al., 2002). To address uncertainty in restoration outcomes, a comprehensive, long-term monitoring programme was implemented for a stream restoration project on the Upper Arkansas River (UAR). This paper evaluates the short-term effectiveness of restoration by investigating changes in habitat quality within 2 years of project implementation.

The primary goals of habitat restoration were to improve brown trout (*Salmo trutta*) population density and biomass and to improve body condition and fish health within the project reach (Stratus Consulting Inc., 2010). Secondary goals were to improve age and size class structure by increasing spawning areas and providing refuge for juvenile trout. Restoration objectives targeted critical habitat functions such as spawning, cover, refuge, and forage production (Schlosser & Angermeier, 1995). Primary monitoring targets included instream habitat structures, riparian vegetation, fish populations, benthic macroinvertebrates, and habitat quality (Stratus Consulting Inc., 2010). The objectives of this study were (1) to evaluate initial changes in habitat quality using a two-dimensional (2D) hydrodynamic modelling approach and (2) to compare multiple metrics of habitat quality. This 2D modelling approach has been successfully applied to quantify changes in fish habitat for other stream restorations (Boavida, Santos, Cortes, Pinheiro, & Ferreira, 2011; Gard, 2014; Koljonen, Huusko, Mäki-Petäys, Louhi, & Muotka, 2013). As fish populations may take five or more years to respond to restoration (Binns, 1994; Hunt, 1976), biological monitoring targets were not evaluated for this initial investigation. Evaluating changes in physical habitat associated with stream restoration will not only support adaptive management for this project but also inform the approach, implementation, and value of future projects.

Habitat quality was quantified using three metrics: weighted usable area (WUA), foraging positions (FP), and habitat heterogeneity. Due to the shortcomings and criticisms of the WUA approach (Railsback, 2016), the impact of stream restoration on FP was evaluated as an alternative measure of habitat quality. Foraging and bioenergetics models indicate that salmonids will select FP that are energetically profitable (Fausch, 1984, 2014). Positions are energetically profitable if they require less energy to occupy than can be gained by foraging. Profitable FP minimize the amount of energy spent on swimming, maintaining position, and searching for prey while maximizing encounter rates for drifting prey (Fausch, 1984; Hayes, Stark, & Shearer, 2000; Hughes, 1998; Hughes, Hayes, & Shearer, 2003). These positions are often associated with complex physical habitat, such as bedform features, changes in substrate size, and large wood. Additionally, increasing habitat heterogeneity has been linked to enhance stream community diversity (Gorman & Karr, 1978; Palmer, Hakenkamp, & Nelson-Baker, 1997) and is a common goal of stream restoration (Palmer, Menninger, & Bernhardt, 2009). Therefore, variability in water depth and velocity was also evaluated as an index of habitat quality. Using multiple lines of evidence to evaluate habitat quality is a unique approach intended to identify strengths, limitations, and comparability of each metric.

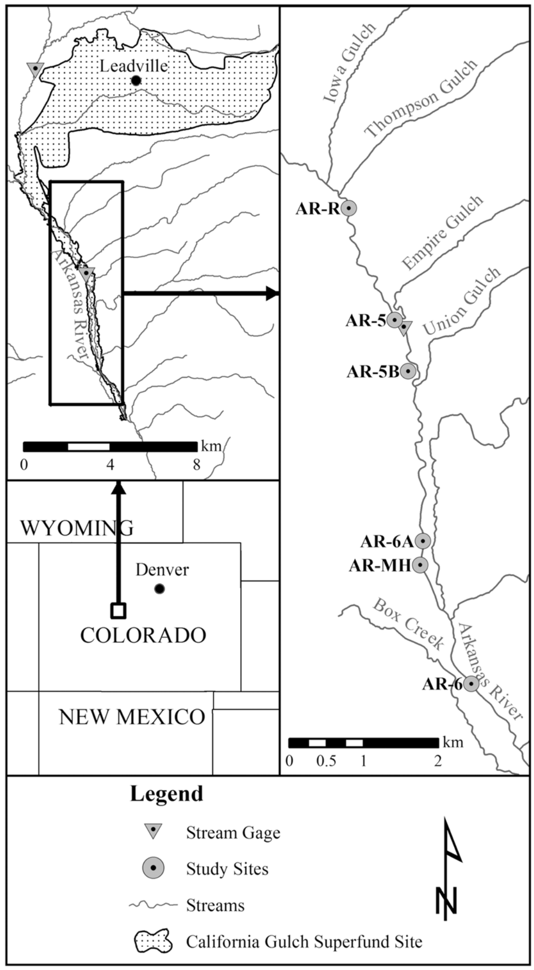

FIGURE 1 Location of study area and study sites. Impact sites: AR-R, AR-5, and AR-MH. Control sites: AR-5B, AR-6, and AR-6A

2 | METHODS

2.1 | Study area

The study site is located in the Southern Rocky Mountains on approximately 8 km of the UAR within the California Gulch Superfund Site near Leadville, Colorado, USA (Figure 1). Historic mining activities, land-use practices, and transmountain water diversions resulted in accelerated bank erosion, channel widening, and degraded habitat diversity (Clements, Vieira, & Church, 2010; Stratus Consulting Inc., 2010). Prior to restoration, the project reach was characterized by an over-wide channel that lacked velocity refuge and deep pools for over-winter habitat (Stratus Consulting Inc., 2010). The primary factor limiting the fishery was metals pollution, but issues with habitat quality and whirling disease had also affected fish density and species diversity, respectively. Due to degraded water quality, brown trout are the only naturally reproducing fish in the UAR and represent more than 75% of the fish community (Policky, 2012, 2015). The study reach is a fourth-order stream in an unconfined alluvial valley with cobble substrate, pool-riffle morphology (Montgomery & Buffington, 1997), and a C3 Rosgen stream classification (Rosgen, 1994). The bedslope for the project reach was 0.64%

and sinuosity was 1.30. The contributing area for the study reach is 692 km^2 with an elevation range of 2,790 to 4,390 m. Hydrology is snowmelt dominated with an average annual precipitation of 0.63 m and an average annual discharge of 4.6 m^3/s.

Water quality remediation projects that addressed point and non-point sources of metals pollutions were implemented prior to habitat restoration. The design of the restoration project had two primary components. First, hydraulic geometry and effective discharge were analysed to develop channel geometries that would transport a range of potential inflowing sediment loads from upstream supply reaches (Hardie, Kulchawik, Beeby, & Bledsoe, 2013). Sensitivity analyses were performed to address uncertainties in existing and future conditions related to inflowing sediment loads, streamflow regime, and hydraulic conditions. Second, habitat and restoration treatments were added in strategic locations to stabilize streambanks, promote diverse stream morphology, reduce erosion and downstream sedimentation, enhance overhead cover for trout, increase spawning areas, and provide refuge for juvenile trout. Treatments included channel narrowing, boulder clusters, boulder and log vanes, point-bar and pool development, sod mats, toe wood, willow transplants, willow plantings, and riparian seeding. Channel narrowing activities were prioritized in areas of high aggradation potential, whereas habitat features that provided velocity refuge, created pools, and enhanced bedform diversity were prioritized in areas of high transport capacity. The location of treated and untreated fluvial tailings deposits was a major design constraint that influenced channel alignment and the need for bank stabilization. Additional information on restoration treatments and quantities was included in Tables S1 and S2.

2.2 | Study design

The effectiveness of habitat restoration was evaluated using a before–after–control–impact study design. Study sites were selected prior to construction to support baseline assessments. The before–after–control–impact design was chosen to account for any natural habitat changes that occurred across control and impact sites during the study period. Three sites were selected to represent restoration areas (AR-R, AR-5, and AR-MH; "impact sites") and three sites represented untreated controls (AR-5B, AR-6, and AR-6A; "control sites"; Figure 1). Rather than using random selection, sites were selected to coincide with locations where fish population monitoring had been conducted prior to restoration. Fish monitoring sites were selected on the basis of accessibility and relative habitat conditions ranging from impaired to reference. Although beyond the scope of this study, results from fish population monitoring will be used to validate changes in habitat quality after five years of post-restoration monitoring has been completed.

Average site length was 191 m with a range of 126–239 m and average channel width was 16.8 m with a range of 10.4–33.2 m. The scale of these sites places them within the segment level (10^2 m) of the stream system (Frissell, Liss, Warren, & Hurley, 1986). Characteristic hydrology was investigated using a U.S. Geological Survey stream gauge within the project reach near Empire Gulch (Figure 1). Average daily discharge data from 1990 to 2015 were used to select flows for habitat analysis by calculating a median discharge value for each day of the water year. Five discharge values were selected to represent a range of flows, including an estimate of the "bankfull" discharge, a "high" flow equivalent to the median yearly maximum flow, an "intermediate" flow, a "low" flow, and a "spawning" flow associated with baseflow conditions that occur during the fall spawning season. As the river splits into two channels above the AR-R site, a unique stage-discharge relationship was developed to estimate flows at this site. Habitat quality at all other study sites was analysed using the same suite of flows. The bankfull, high, intermediate, low, and spawning flows for AR-R were 12.5, 10.0, 4.7, 2.5, and 1.3 m^3/s, respectively, compared with 20.7, 16.4, 7.6, 4.0, and 2.0 m^3/s, respectively, for all other study sites.

2.3 | Site surveys

All sites were surveyed in 2013 to evaluate baseline conditions prior to restoration. Instream construction occurred during summer and fall months in 2013 and 2014. All sites were re-surveyed in 2014 following construction and again in 2016 to evaluate changes following two run-off cycles. Topographic surveys were conducted with survey-grade GPS using NAD 1983 U.S. State Plate Central and NAVD 1988 coordinate systems and then re-projected into UTM NAD 1983 13N to support model configuration. Five passes, or breaklines, were surveyed within the active channel along longitudinal slope breaks to characterize streambed morphology. If defined slope breaks were not evident, breaklines were equally spaced between bank bottoms. Survey points were collected every 3–5 m and at all major changes in slope or geometry. Breaklines were also surveyed along the top and bottom of each bank, the adjacent floodplain, and around any islands. Water surface elevations were surveyed along each bank to support calibration of hydraulic models. Substrate type was also surveyed to develop channel index files using the following sediment classes: clay, silt (<0.062 mm), sand (0.062–2 mm), gravel (2–64 mm), cobble (64–250 mm), boulder (>250 mm), and bedrock. Representative pebble counts (Rosgen, 2008) and discharge measurements were conducted during each survey.

2.4 | Habitat modelling

Habitat modelling was conducted with River2D Version 0.95, a 2D, depth-averaged model of river hydrodynamics and fish habitat (Steffler & Blackburn, 2002; Waddle & Steffler, 2002). One-dimensional hydraulic models were created in HEC-RAS v4.1 (Brunner, 2010) to estimate boundary conditions for 2D models. HEC-RAS models were calibrated by varying Manning's n to minimize the difference between surveyed and modelled water surface elevations. Following calibration, steady-state flow analyses were performed in HEC-RAS to estimate upstream and downstream water surface elevations for selected flows.

Survey data were used to create a finite element mesh to represent channel morphology in River2D Mesh (Waddle & Steffler, 2002). Each mesh was initially developed using a uniform fill with 1.0 m spacing.

Breaklines were added at bank tops and bottoms to reduce discretization error. Additional nodes were added in areas with more geomorphic complexity. River2D models were calibrated by iteratively changing the effective roughness height (k_s) to minimize the difference between surveyed and modelled water surface elevations. The number of water surface points used for model calibration ranged from 36 to 185 per site with an average of 103 over the study period. The average difference between surveyed and modelled water surface elevations was 9.8 mm, with a range from −25 to 28 mm. The range of flows used for model calibration was 1.38–3.46 m^3/s across all study sites and years, with an average of 2.61 m^3/s. Calibration resulted in close agreement between inflow and outflow discharges with differences less than 1%.

Hydraulic model outputs were used to calculate WUA by applying habitat suitability curves (HSC) derived from observations of brown trout behaviour (Raleigh, Zuckerman, & Nelson, 1986). HSC for depth, velocity, and substrate were obtained for brown trout adult, juvenile, fry, and spawning life stages (Figure 2). Juvenile and fry HSC were taken from Raleigh et al. (1986); adult HSC were obtained from Ayllón, Almodóvar, Nicola, and Elvira (2010); and spawning HSC were taken from Louhi, Mäki-Petäys, and Erkinaro (2008). Suitability indices for depth, velocity, and channel substrate were aggregated into a combined suitability index (CbSI). Total WUA was calculated for each life stage and flow, with the exception of WUA for spawning fish, which was estimated for the spawning discharge only. WUA represents the spatial summation of total area weighted by the CbSI for each life stage, site, and year. Estimates of WUA were constrained between the as-built (2014) bank tops to focus on instream habitat and normalized by site length to support comparisons across sites.

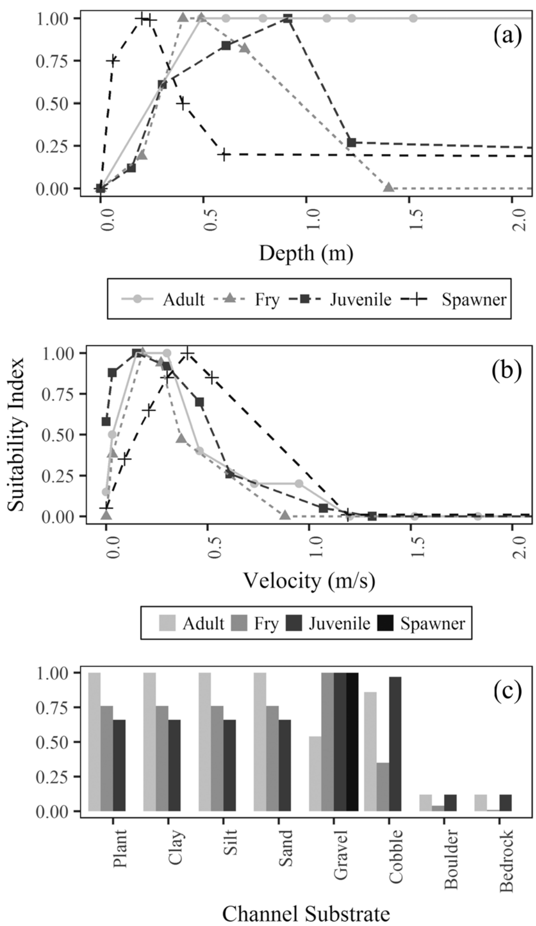

FIGURE 2 Habitat suitability curves for (a) depth, (b) velocity, and (c) channel substrate used to model brown trout (*Salmo trutta*) habitat for adult, fry, juvenile, and spawning life stages

2.5 | Foraging positions

Foraging positions were defined as any location that met a combination of velocity, proximity, and depth criteria based on salmonid drift-foraging geometry and bioenergetics models (Hayes et al., 2000; Hayes & Jowett, 1994; Hughes, 1998; Hughes et al., 2003). In these models, a focal point provides a resting position where a fish can search for prey in adjacent capture areas. Capture areas occur within a foraging radius that represents the reaction distance over which a fish can capture drifting prey. The criteria used to evaluate FP were presented in Table 1. Fish lengths and weights were obtained from electrofishing surveys at study sites. The median fish length from these surveys was used to analyse foraging positions for adult brown trout. Depth criteria were based on the minimum foraging depths reported by Grant (1999) and Hughes et al. (2003). As fish can forage vertically and horizontally, depth-average velocities were extrapolated using the vertical-velocity curve method to evaluate vertical FP (Buchanan & Somers, 1969). Vertical-velocity curves were developed for each site assuming a logarithmic relationship for a turbulent velocity profile. To compare results across sites, the number of FP was normalized by site length. This study identifies FP that were profitable for a fish from a hydraulic perspective only. Other factors that influence FP, such as competition, drifting prey density, and water quality, were beyond the scope of this study.

2.6 | Habitat heterogeneity

To quantify restoration impacts on morphological heterogeneity, we evaluated variability in channel hydraulics by calculating coefficient of variation (CV) for water depth and velocity, as done by previous

TABLE 1 Variables used to evaluate foraging positions within the Upper Arkansas River stream restoration project

Variable	Value
Fish length (mm)	202 ± 3.5
Fish weight (g)	272 ± 75
Focal point velocity (m/s)	0.15 − 0.20
Capture area velocity (m/s)	0.27 − 0.33
Foraging radius (m)	1.0
Minimum depth (m)	0.3

Note. Median values were given for fish length and weight followed by standard deviation. References for selected values were as follows: focal point velocity (Hayes et al., 2000; Hayes & Jowett, 1994), capture area velocity (Hayes & Jowett, 1994; Hayes et al., 2000; Stewart et al., 1981), foraging radius (Hayes & Jowett, 1994), and minimum depth (Grant, 1999; Hughes et al., 2003).

studies (Palmer et al., 1997; Palmer et al., 2009; Pretty et al., 2003). Values for depth, velocity, depth suitability index (DSI), and velocity suitability index (VSI) values were taken from each cell of the River2D model. Means for depth, velocity, DSI, and VSI and the CV for depth and velocity were calculated from these cell values. The number of cells per site ranged from 7,869 to 18,111 with an average of 13,226. Changes in WUA and FP were informed by analysing the mean depth, velocity, DSI, and VSI. Although DSI and VSI were tied to a specific species and life stage, depth and velocity represent hydraulic conditions that were independent of fish biology.

2.7 | Statistical analysis

Data were analysed using analysis of variance (ANOVA) of linear models in the R Statistical Package (R Development Core Team, 2017). Linear models were developed for all sites, impact sites, and control sites using WUA and FP as response variables. Spawning WUA was evaluated separately and excluded from models of WUA for adults, juveniles, and fry. Models for WUA included life stage, discharge, and year as explanatory variables, whereas those for FP included discharge and year. Spawning WUA was evaluated using year as the lone explanatory variable. Significant changes in habitat heterogeneity and hydraulics were determined via ANOVA using the respective CV or mean as the response variable and site and year as explanatory variables. Due to vastly mismatching variances, separate models for control and impact sites were built in addition to full models. An α of .05 was used to identify significant differences and interactions. Mean separation via pairwise comparisons using the "lsmeans" package (Lenth, 2016) was performed when significant differences or interactions were found. Where least square mean values were reported, the least square mean has been averaged across all other model parameters. Detailed results for all statistical analyses were reported in Appendices S1 to S4.

3 | RESULTS

3.1 | Habitat modelling

Individual control sites exhibited increases and decreases in WUA during the study period, whereas WUA at impact sites tended to increase from 2013 to 2014 and then decreased from 2014 to 2016 (Figure 3). Averaged across years, discharge, and life stage, control sites had 18.5% greater WUA than impact sites ($P < .001$; Figure 4). The WUA at impact sites increased (12.2%) between 2013 and 2014 ($P = 0.042$). The initial increase in WUA at impact sites was followed by a decrease (10.2%) from 2014 to 2016 ($P = 0.084$; Figure 4). Control sites exhibited no change in WUA from 2013 to 2014 ($P = 0.873$) or 2014 to 2016 ($P = 0.783$). No significant interactions between year and discharge or year and life stage were observed at control ($P = 0.999$ and $P = 0.970$, respectively) or impact sites ($P = 0.980$ and $P = 0.685$, respectively). For impact sites, WUA for spawning brown trout was similar from 2013 to 2014 but increased by 53.3% from 2014 to 2016 ($P = 0.007$), whereas control sites

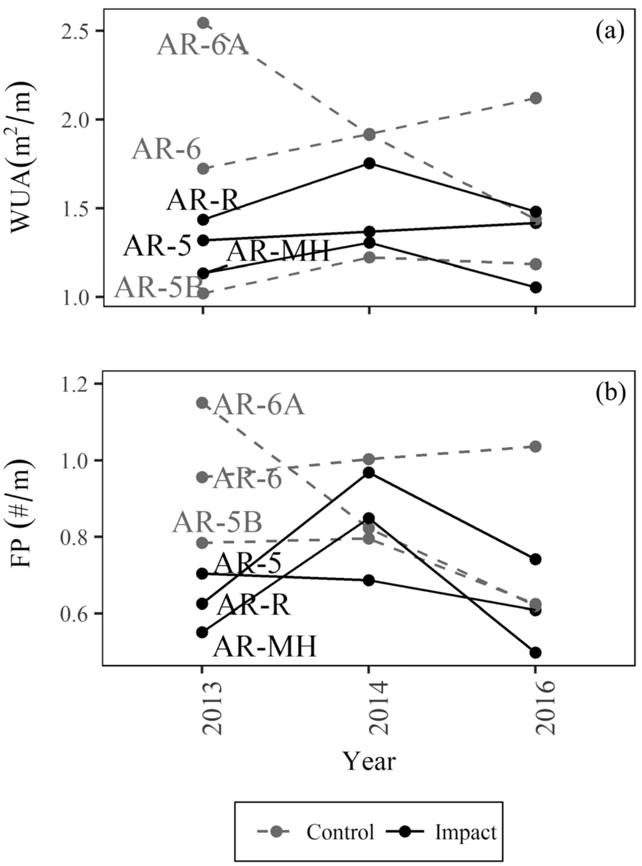

FIGURE 3 Estimates of (a) weighted usable area (WUA) and (b) foraging positions (FP) for individual study sites by year

did not exhibit any significant changes in spawning WUA across years ($P = 0.997$; Figure 4).

Discharge and life stage interacted in their effects on WUA at both impact and control sites ($P < .001$ and $P = 0.026$, respectively). Generally, WUA decreased as discharge increased for all life stages (Figure 5). At impact sites, adult WUA decreased 59.8% from spawning to bankfull discharges ($P < .001$), whereas juvenile WUA decreased 68.8% ($P < .001$). At impact sites, the adult WUA was generally greater than juvenile WUA ($P < .001$). Fry, which tended to have lower WUA that adults or juveniles ($P < .001$ and $P < .001$, respectively), had similar WUA across all discharges (Figure 5). At control sites, similar trends in WUA due to the discharge by life stage interaction were observed. From spawning to bankfull discharges, WUA decreased by 53.0% for adults and 70.1% for juveniles. At spawning, low and intermediate discharges, adults and juveniles had similarly high WUA at control sites. For adults and juveniles, high and bankfull discharges resulted in similarly low estimates of WUA. At control sites, adult WUA tended to be higher than WUA for juveniles ($P = 0.037$), whereas fry had lower WUA than either adults or juveniles ($P < .001$ and $P < .001$, respectively).

No significant changes in average depth or DSI were observed at control or impact sites (Table 2). Water velocities, whether measured as mean velocity or the VSI, did not change during the study period. Complete ANOVA results for analysis of WUA, DSI, and VSI were

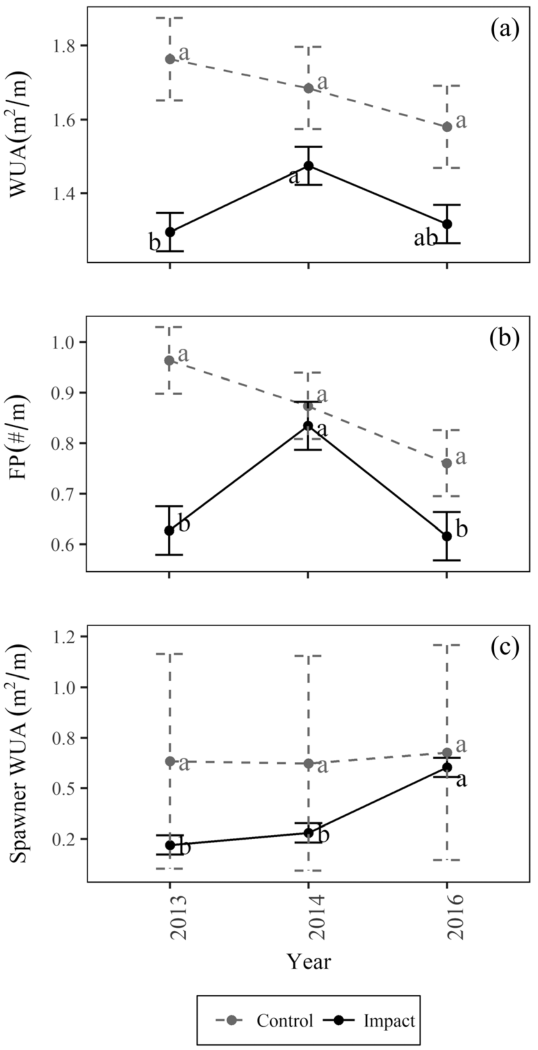

FIGURE 4 Least-square mean estimates and standard error for (a) weighted usable area (WUA), (b) foraging positions (FP), and (c) WUA for spawning fish at control and impact sites by study years averaged over study discharges and life stages. Differing lowercase letters indicate differences of $P < .05$. Grey lowercase letters indicate differences for control sites, whereas black lowercase letters indicate differences for impact sites. Lowercase letters do not indicate differences between control and impact sites

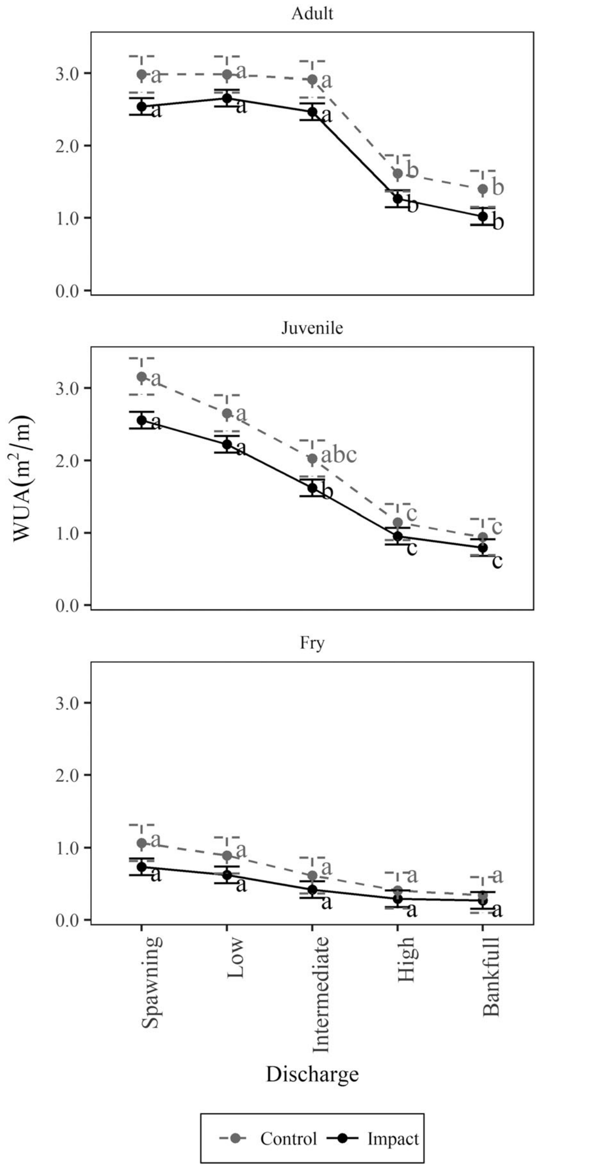

FIGURE 5 Least-square mean estimates and standard error for weighted usable area (WUA) at control and impact sites by discharges and life stages averaged over study years. Discharge by life stage interaction is significant (impact sites: $P < .001$; control sites: $P = 0.023$). Differing lowercase letters indicate differences of $P < .05$. Grey lowercase letters indicate differences for control sites, whereas black lowercase letters indicate differences for impact sites. Lowercase letters do not indicate differences between control and impact sites

presented in Appendices S1 and S2. Images of the CbSI for all sites, years, life stages, and flows were included in Appendix S5.

3.2 | Foraging positions

Similar to WUA, FP for adult trout at individual control sites both increased and decreased during the study period, whereas impact sites generally exhibited an increase in FP from 2013 to 2014 followed by a decline from 2014 to 2016 (Figure 3). Averaged across year and discharge, control sites had 20.0% more FP than impact sites (0.865/m vs. 0.692/m; $P < .001$). At impact sites, FP increased by 24.8% from 2013 to 2014 and decreased by 26.1% from 2014 to 2016 (Figure 4). The number of FP at control sites did not vary across years. No interaction between year and discharge was observed at either control or impact sites ($P = 0.842$ and $P = 0.947$, respectively). As discharge increased, FP decreased at both control and impact sites (Figure 6). Complete ANOVA tables for analysis of FP were included in Appendix S4.

3.3 | Habitat heterogeneity

Habitat heterogeneity rarely differed between control and impact sites or among years (Table 2). The only significant change in habitat

TABLE 2 Least-square mean estimates of mean (M), coefficient of variation (CV), and standard error (SE) for water depth and velocity and mean and SE for depth suitability index (DSI) and velocity suitability index (VSI) at control (n = 3) and impact sites (n = 3)

		Depth (m)				DSI		Velocity (m/s)				VSI	
Site	Year	*M*	±*SE*	CV	±*SE*	*M*	±*SE*	*M*	±*SE*	CV	±*SE*	*M*	±*SE*
Impact	2013	0.43_a	0.03	0.52_b	0.02	0.60_a	0.04	0.85_a	0.06	0.59_a	0.03	0.19_a	0.02
	2014	0.46_a	0.03	0.58_{ab}	0.02	0.52_a	0.04	0.79_a	0.06	0.62_a	0.03	0.21_a	0.02
	2016	0.43_a	0.03	0.60_a	0.02	0.50_a	0.04	0.86_a	0.06	0.58_a	0.03	0.21_a	0.02
Control	2013	0.54_a	0.03	0.58_a	0.02	0.53_a	0.03	0.77_a	0.06	0.63_a	0.03	0.23_a	0.02
	2014	0.51_a	0.03	0.58_a	0.03	0.54_a	0.03	0.79_a	0.06	0.61_a	0.03	0.23_a	0.02
	2016	0.48_a	0.04	0.61_a	0.03	0.53_a	0.03	0.84_a	0.06	0.64_a	0.03	0.22_a	0.02

Note. For each year, depth and velocity were summarized across site and discharge (n = 15), whereas DSI and VSI were summarized across site, discharge, and life stage (n = 48). Differing lowercase letters indicate differences of $P < .05$ within impact sites or within control sites. Lowercase letters do not indicate differences between control and impact sites.

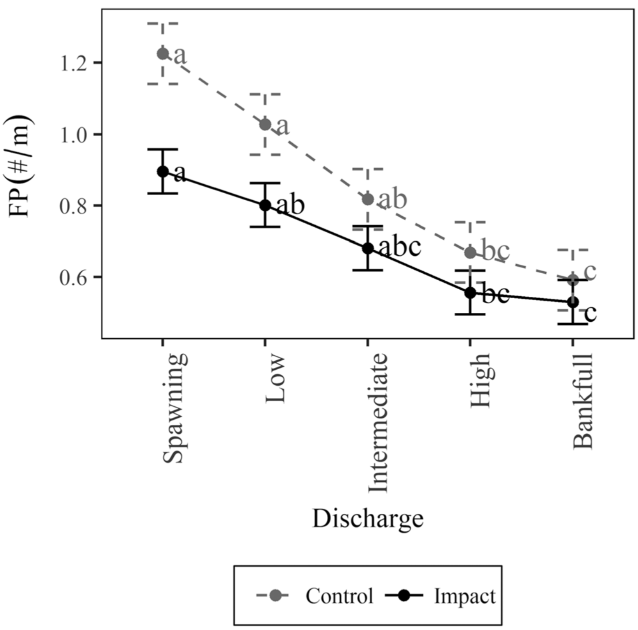

FIGURE 6 Least-square mean estimates and standard error for foraging positions (FP) at control and impact sites across study discharges averaged over study years. Differing lowercase letters indicate differences of $P < .05$. Grey lowercase letters indicate differences for control sites, whereas black lowercase letters indicate differences for impact sites. Lowercase letters do not indicate differences between control and impact sites

heterogeneity was the CV for depth at impact sites, which increased by 15.4% from 2013 to 2016 ($P = 0.046$). The CV for depth did not change at control sites over the study period. No changes in the CV for velocity were observed at either control or impact sites (Table 2). ANOVA tables for CV analysis were presented in Appendix S3.

4 | DISCUSSION

Results indicated that stream restoration had an ambiguous impact on habitat quality. Although WUA and FP both increased following restoration, this initial improvement was followed by declines to pre-restoration levels. However, analyses based on WUA alone may not have adequately captured habitat changes due to a "net change" effect. Prior to restoration, habitat of moderate quality existed across impact sites (Figure 7; Appendix S5). Restoration treatments created areas of high habitat quality adjacent to areas of lower quality. This resulted to no net change in WUA, suggesting that restoration had no impact. In reality, restoration created patches of both very high and low quality habitat, which could be more beneficial for fish populations than homogenously mediocre habitat.

The lack of sustained change in WUA reported in this study may also be related to the scale of analysis. Although this analysis is focused on impacts at the stream segment scale, the restoration project was implemented over a much larger scale and individual treatments were designed to affect habitat quality at smaller scales. Restoration activities occurring at the larger, stream level (10^3 m) scale could impact sediment transport capacity and supply in upstream segments, which could affect habitat quality within study sites. Furthermore, evaluating habitat quality at the segment level may not capture the effects of restoration activities at smaller scales, such as reach (10^1 m), pool/riffle (10^0 m), or microhabitat (10^{-1} m) levels (Frissell et al., 1986). All control sites were located within (AR-5B and AR-6A) or downstream (AR-6) of the project reach. As such, control sites could have been affected by stream level activities occurring in adjacent segments but should not have been affected by reach or smaller scale changes associated with individual restoration treatments.

Our results indicate that control sites had higher habitat quality than impact sites, but this was likely an artefact of site selection and should not be interpreted as an impact of restoration activities or environmental conditions. The overall goal of the project was to improve instream trout habitat. As such, degraded sites were targeted for restoration, whereas control sites were selected to represent various habitat conditions ranging from impaired to functioning. If monitoring sites were selected in a truly random manner, some degraded sites would have been left untreated, whereas less degraded sites were restored. However, a randomized approach to site selection could undermine project goals and squander project resources. The project was also limited by the number of monitoring sites that could be established. Given the wide variation in habitat quality between the sites, the addition of more monitoring sites would have been valuable.

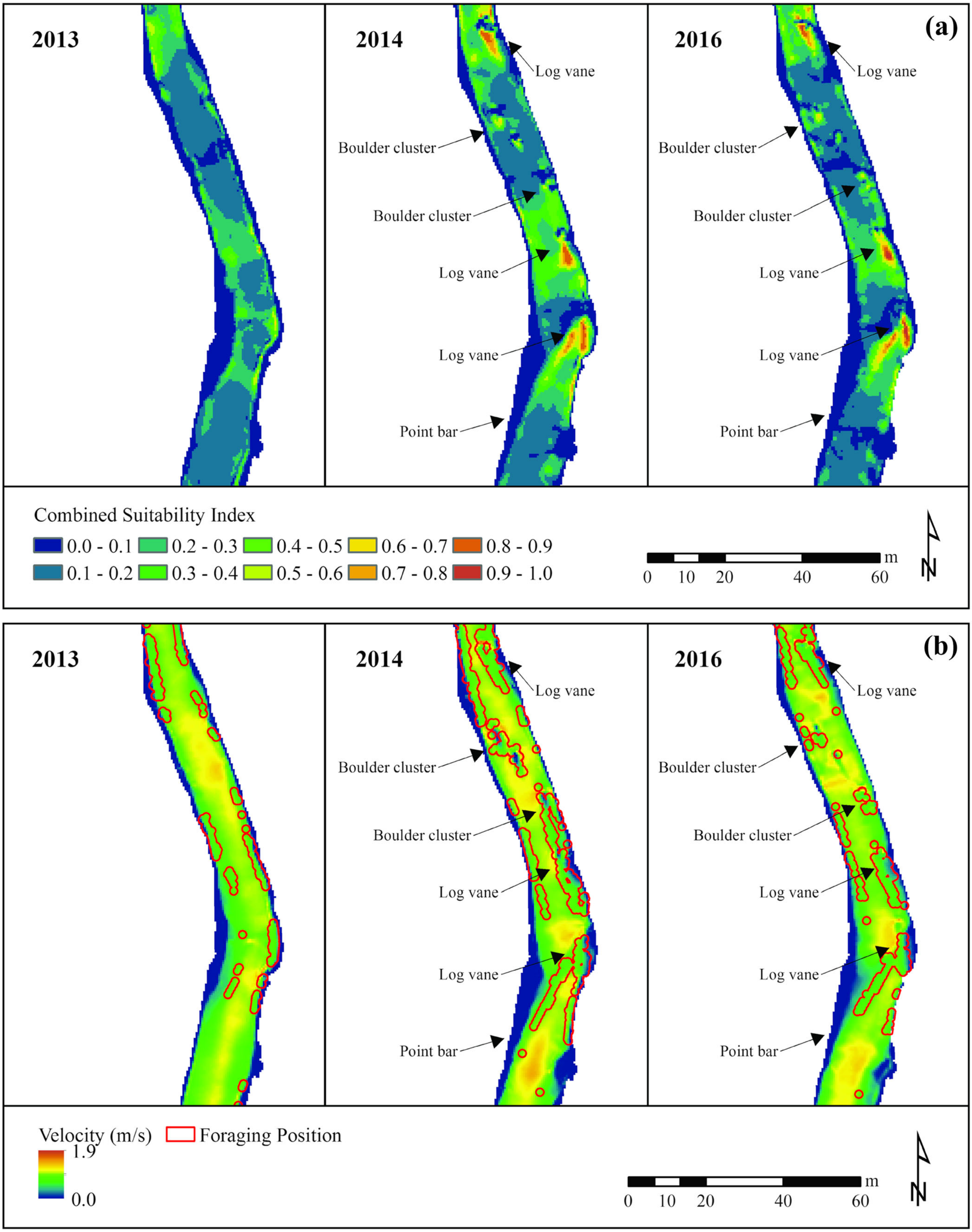

FIGURE 7 Spatial depiction of (a) weighted usable area (as combined suitability index) at 1.3 m^3/s and (b) foraging positions at 2.5 m^3/s for adult brown trout at impact site AR-R

Increasing the number of study sites would also support the inclusion of relative habitat condition prior to restoration (e.g., impaired, functioning-at-risk, or reference) as a factor when analysing changes in habitat quality.

The approach to habitat modelling presented in the paper has limitations. Specifically, HSC have been criticized as out of date and potentially erroneous (Railsback, 2016). Calculations of WUA apply equal weighting to depth, velocity, and channel substrate as predictors of habitat, but the relative importance of each variable may vary. Additionally, other habitat variables, such as cover, may have an equal or greater impact on habitat quality (Railsback, 2016; Wesche, Goertler, & Frye, 1987; Wesche, Goertler, & Hubert, 1987). The 2D models used in this study cannot resolve the presence of undercut banks, which contribute to habitat quality by providing refuge, cover, and FP. Our analysis of FP also neglected several factors that affect foraging behaviour, such as prey type, life history changes, and availability of cover (Grant, 1999; Hughes, 1998). Furthermore, this study focused solely on instream habitat and did not evaluate changes in floodplain connectivity or overbank habitat. Inundated floodplains can provide important habitat for all life stages of fish when discharge is high (King, Humphries, & Lake, 2003). Although the core strength of WUA is an index of potential habitat changes with flow (Reiser & Hilgert, 2018), there is no exact inference mechanism between WUA and quantifiable fisheries metrics. Validating habitat quality results by investigating changes in fish populations, documenting the location of spawning redds, or observing actual foraging behaviour at study sites would help determine the utility of these habitat-modelling approaches.

The significant changes in WUA and FP observed at impact sites were likely attributed to changes in channel hydraulics associated with habitat treatments. Although changes were not statistically significant, impact sites exhibited increases in average depth and decreases in average velocity from 2013 to 2014. This suggests that the significant increase in habitat quality at impact sites was related to treatments that provided velocity refuge and deeper pools. The subsequent decline in WUA from 2014 to 2016 was associated with a decrease in average depth, likely due to the filling of developed pools. For example, the initial improvements at impact site AR-R were associated with pool development near log vanes, boulder clusters, and point-bar development (Figure 7; Appendix S5). Improved habitat was evident at developed pools downstream of log vanes and velocity refuges behind boulder clusters. Declines in habitat quality between 2014 and 2016 also coincide with these locations, indicating that changes in habitat metrics were associated with restoration treatments rather than extraneous environmental influences. Furthermore, significant changes in habitat metrics were not observed at control sites, indicating that the changes at impact sites were due to restoration activities.

Changes in WUA and FP indicate that habitat was enhanced following restoration, but those changes were not sustained. However, images of the CbSI at impact sites suggest more patches of high-quality habitat were present in 2016 compared with pre-restoration conditions in 2013 (Figure 7; Appendix S5). Furthermore, the increased CV for depth at impact sites from 2013 to 2016 indicates that restoration had beneficial impacts on habitat heterogeneity that were sustained. In contrast, the CV for depth at control sites yielded no uniform trends. This, again, indicates that the similar patterns observed at impacts sites were likely an effect of habitat restoration and not environmental conditions. Pretty et al. (2003) found that sites treated with instream structures that had increased CV for depth and velocity experienced subsequent improvements in fish populations, indicating that improving these metrics is associated with improved habitat quality. However, the increase in depth variability indicates that the project reach could still be evolving in response to restoration treatments. In which case, additional monitoring would be needed to document long-term changes in habitat quality.

An increase in spawning habitat was observed between 2014 and 2016. Instream structures designed to create or maintain pools can facilitate the deposition and storage of spawning gravels (House & Boehne, 1986). As no increases in spawning habitat were observed at control sites, it is likely that the combination of channel narrowing and habitat structures increased sediment transport capacity and facilitated the deposition of gravels at impact sites. The increase in spawning WUA at impact sites could also be associated with high flow events during the study period. Habitat dynamics at both control and impact sites can be partially attributed to the 6-, 8- and 3-year flood events that occurred from 2014 to 2016, respectively. Flows of these magnitudes can induce channel maintenance functions, including mobilization of bedload sediment, scour of vegetation from the channel, inundation of floodplains, lateral channel migration, and reshaped alluvial features (Schmidt & Potyondy, 2004).

At impact sites, mobilized sediment was deposited in developed pools, resulting in the observed loss of depth and associated declines in WUA between 2014 and 2016. Although the D_{50} sediment size increased by 32% between 2014 and 2016 across all sites, the D_{50} remained classified as small cobble (64–90 mm) for all study years. The apparent increase in the D_{50} could be indicative of coarse sediment deposition and flushing of fine sediment, both of which would be expected during channel maintenance events. The bedform diversity initially created through the construction of lateral scour pools, vanes, and boulder clusters subsequently declined as pools filled in with sediment during post-construction run-off events. Investigating structure performance associated with maintenance of desirable pool depths in the presence of high sediment supply would help inform future restoration designs. Implementation of the restoration project could have been phased so that channel narrowing and bank stabilization were conducted first. Habitat treatments that addressed bedform diversity and velocity refuge could then be added during a second phase once the channel had adjusted to a new dynamic equilibrium.

Significant declines in WUA and FP observed from 2014 to 2016 indicate that the long-term effects of stream restoration on trout habitat in the UAR remains uncertain. Although the initial improvements in WUA and FP were not sustained, improved spawning habitat and depth variability at impact sites suggests habitat quality was enhanced, which highlights the value of using multiple lines of evidence. Fish population monitoring may provide additional insight regarding the effectiveness of this restoration project. Due to the relatively short study period, habitat conditions could still be evolving. Wohl et al.

(2005) stresses the dynamic nature of river systems, indicating that expectations for river restoration projects to achieve static conditions or fixed end points are inappropriate. Rivers should be expected to adapt and respond to changing conditions, such as those induced by restoration. This suggests that more passive and adaptive approaches to restoration that target processes such as sediment transport and reestablishment of riparian vegetation could be more effective and sustainable than projects that rely on the intensive use of instream habitat structures. Future research will compare changes in habitat quality and fishery metrics to evaluate the biological effectiveness of this restoration project. The results from this study indicate that comprehensive and long-term monitoring is critically important for informing adaptive management and improving the effectiveness of stream restoration projects.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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REFERENCES

Ayllón, D., Almodóvar, A., Nicola, G. G., & Elvira, B. (2010). Ontogenetic and spatial variations in brown trout habitat selection. *Ecology of Freshwater Fish*, *19*, 420–432. https://doi.org/10.1111/j.1600-0633.2010.00426.x

Bash, J. S., & Ryan, C. M. (2002). Stream restoration and enhancement projects: Is anyone monitoring? *Environmental Management*, *29*, 877–885. https://doi.org/10.1007/s00267-001-0066-3

Bennett, S., Pess, G., Bouwes, N., Roni, P., Bilby, R. E., Gallagher, S., … Greene, C. (2016). Progress and challenges of testing the effectiveness of stream restoration in the Pacific Northwest using intensively monitored watersheds. *Fisheries*, *41*, 93–103.

Bernhardt, E. S., Sudduth, E. B., Palmer, M. A., Allan, J. D., Meyer, J. L., Alexander, G., … Pagano, L. (2007). Restoring rivers one reach at a time: Results from a survey of U.S. river restoration practitioners. *Restoration Ecology*, *15*, 482–493. https://doi.org/10.1111/j.1526-100X.2007.00244.x

Binns, N. A. (1994). Long-term responses of trout and macrohabitats to habitat management in a Wyoming headwater stream. *North American Journal of Fisheries Management*, *14*, 87–98. https://doi.org/10.1577/1548-8675(1994)014<0087:LTROTA>2.3.CO;2

Boavida, I., Santos, J. M., Cortes, R., Pinheiro, A., & Ferreira, M. T. (2011). Benchmarking river habitat improvement. *River Research and Applications*, *28*, 1768–1779.

Brunner, G. W. (2010). Hydrologic Engineering Center's River Analysis System (HEC-RAS). US Army Corps of Engineers. Retrieved from http://www.hec.usace.army.mil/software/hec-ras/

Buchanan, T. J., & Somers, W. P. (1969). Discharge measurement at gaging stations. In *Techniques of water-resources investigations of the United States Geological Survey* (p. 65). Washington, DC: United States Government Printing Office.

Clements, W. H., Vieira, N. K. M., & Church, S. E. (2010). Quantifying restoration success and recovery in a metal-polluted stream: A 17-year assessment of physicochemical and biological responses. *Journal of Applied Ecology*, *47*, 899–910. https://doi.org/10.1111/j.1365-2664.2010.01838.x

Fausch, K. D. (1984). Profitable stream positions for salmonids: Relating specific growth rate to net energy gain. *Canadian Journal of Zoology*, *62*, 441–451. https://doi.org/10.1139/z84-067

Fausch, K. D. (2014). A historical perspective on drift foraging models for stream salmonids. *Environmental Biology of Fishes*, *97*, 453–464. https://doi.org/10.1007/s10641-013-0187-6

Frissell, C. A., Liss, W. J., Warren, C. E., & Hurley, M. D. (1986). A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. *Environmental Management*, *10*, 199–214. https://doi.org/10.1007/BF01867358

Gard, M. (2014). Modelling changes in salmon habitat associated with river channel restoration and flow-induced channel alterations. *River Research and Applications*, *30*, 40–44. https://doi.org/10.1002/rra.2642

Gorman, O. T., & Karr, J. R. (1978). Habitat structure and stream fish communities. *Ecology*, *59*, 507–515. https://doi.org/10.2307/1936581

Grant, G. C. (1999). *Growth, spawning behavior and feeding strategies of trout in a small midwestern stream (doctoral dissertation)*. Minneapolis: University of Minnesota.

Hardie, K., Kulchawik, P., Beeby, J., & Bledsoe, B. P. (2013). *Rehabilitation of the Hayden and Reddy reaches of the upper Arkansas River NRD*. Fort Collins, CO: Colorado State University.

Hayes, J. W., & Jowett, I. G. (1994). Microhabitat models of large drift-feeding brown trout in three New Zealand rivers. *North American Journal of Fisheries Management*, *14*, 710–725. https://doi.org/10.1577/1548-8675(1994)014<0710:MMOLDF>2.3.CO;2

Hayes, J. W., Stark, J. D., & Shearer, K. A. (2000). Development and test of a whole-lifetime foraging and bioenergetics growth model for drift feeding brown trout. *Transactions of the American Fisheries Society*, *129*, 315–332. https://doi.org/10.1577/1548-8659(2000)129<0315:DATOAW>2.0.CO;2

House, R. A., & Boehne, P. L. (1986). Effects of instream structures on salmonid habitat and populations in Tobe Creek, Oregon. *North American Journal of Fisheries Management*, *6*, 38–46. https://doi.org/10.1577/1548-8659(1986)6<38:EOISOS>2.0.CO;2

Hughes, N. F. (1998). A model of habitat selection by drift-feeding stream salmonids at different scales. *Ecology*, *79*, 281–294. https://doi.org/10.1890/0012-9658(1998)079[0281:AMOHSB]2.0.CO;2

Hughes, N. F., Hayes, J. W., & Shearer, K. A. (2003). Testing a model of drift-feeding using three-dimensional videography of wild brown trout, *Salmo trutta*, in a New Zealand river. *Canadian Journal of Fisheries and Aquatic Sciences*, *60*, 1462–1476. https://doi.org/10.1139/f03-126

Hunt, R. L. (1976). A long-term evaluation of trout habitat development and its relation to improving management-related research. *Transactions of the American Fisheries Society*, *105*, 361–364. https://doi.org/10.1577/1548-8659(1976)105<361:ALEOTH>2.0.CO;2

Kauffman, J. B., Beschta, R. L., Otting, N., & Lytjen, D. (1997). An ecological perspective of riparian and stream restoration in the western United States. *Fisheries*, *22*, 12–24. https://doi.org/10.1577/1548-8446(1997)022<0012:AEPORA>2.0.CO;2

King, A. J., Humphries, P., & Lake, P. S. (2003). Fish recruitment on floodplains: The roles of patterns of flooding and life history characteristics. *Canadian Journal of Fisheries and Aquatic Sciences*, *60*, 773–786. https://doi.org/10.1139/f03-057

Koljonen, S., Huusko, A., Mäki-Petäys, A., Louhi, P., & Muotka, T. (2013). Assessing habitat suitability for juvenile Atlantic salmon in relation to in-stream restoration and discharge variability. *Restoration Ecology*, *21*, 344–352. https://doi.org/10.1111/j.1526-100X.2012.00908.x

Lenth, R. V. (2016). Least-squares means: The R package lsmeans. *Journal of Statistical Software*, *69*, 1–33.

Louhi, P., Mäki-Petäys, A., & Erkinaro, J. (2008). Spawning habitat of Atlantic salmon and brown trout: General criteria and intragravel factors. *River Research and Applications*, *24*, 330–339. https://doi.org/10.1002/rra.1072

Miller, J. R., & Kochel, R. C. (2009). Assessment of channel dynamics, in-stream structures and post-project channel adjustments in North Carolina and its implications to effective stream restoration. *Environmental Earth Sciences*, *59*, 1681–1692.

Miller, J. R., & Kochel, R. C. (2012). Use and performance of in-stream structures for river restoration: A case study from North Carolina. *Environmental Earth Sciences*, *68*, 1563–1574.

Montgomery, D. R., & Buffington, J. M. (1997). Channel-reach morphology in mountain drainage basins. *Geological Society of America Bulletin*, *109*, 596–611. https://doi.org/10.1130/0016-7606(1997)109<0596:CRMIMD>2.3.CO;2

Palmer, M. A., Hakenkamp, C. C., & Nelson-Baker, K. (1997). Ecological heterogeneity in streams: Why variance matters. *Journal of the North American Benthological Society*, *16*, 189–202. https://doi.org/10.2307/1468251

Palmer, M. A., Menninger, H. L., & Bernhardt, E. (2009). River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice. *Freshwater Biology*, *55*, 1–18.

Pierce, R., Podner, C., & Carim, K. (2013). Response of wild trout to stream restoration over two decades in the Blackfoot River Basin, Montana. *Transactions of the American Fisheries Society*, *142*, 68–81. https://doi.org/10.1080/00028487.2012.720626

Policky, G. (2012). *Fisheries inventories Upper Arkansas River Basin*. Salida, CO: Colorado Parks and Wildlife.

Policky, G. (2015). *Fisheries inventories Upper Arkansas River Basin*. Salida, CO: Colorado Parks and Wildlife.

Pretty, J. L., Harrison, S. S. C., Shepherd, D. J., Smith, C., Hildrew, A. G., & Hey, R. D. (2003). River rehabilitation and fish populations: Assessing the benefit of instream structures. *Journal of Applied Ecology*, *40*, 251–265. https://doi.org/10.1046/j.1365-2664.2003.00808.x

R Development Core Team. (2017). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Railsback, S. F. (2016). Why it is time to put PHABSIM out to pasture. *Fisheries*, *41*, 720–725. https://doi.org/10.1080/03632415.2016.1245991

Raleigh, R. B., Zuckerman, L. D., & Nelson, P. C. (1986). *Habitat suitability index models and instream flow suitability curves: Brown trout*. Washington, DC: U.S. Fish and Wildlife Service.

Reiser, D. W., & Hilgert, P. J. (2018). A practitioner's perspective on the continuing technical merits of PHABSIM. *Fisheries*, *43*, 278–283. https://doi.org/10.1002/fsh.10082

Roni, P., Beechie, T. J., Bilby, R. E., Leonetti, F. E., Pollock, M. M., & Pess, G. R. (2002). A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management*, *22*, 1–20. https://doi.org/10.1577/1548-8675(2002)022<0001:AROSRT>2.0.CO;2

Rosgen, D. L. (1994). A classification of natural rivers. *Catena*, *22*, 169–199. https://doi.org/10.1016/0341-8162(94)90001-9

Rosgen, D. L. (2008). *River stability field guide*. Fort Collins, CO: Wildland Hydrology.

Schlosser, I. J., & Angermeier, P. L. (1995). Spatial variation in demographic processes of lotic fishes: Conceptual models, empirical evidence, and implications for conservations. *Evolution and the aquatic ecosystems: Defining unique units in population conservation*, 17. American Fisheries Society Symposium, 392–401.

Schmetterling, D. A., & Pierce, R. W. (1999). Success of instream habitat structures after a 50-year flood in Gold Creek, Montana. *Restoration Ecology*, *7*, 369–375. https://doi.org/10.1046/j.1526-100X.1999.72031.x

Schmidt, L. J., & Potyondy, J. P. (2004). *Quantifying channel maintenance instream flows: An approach for gravel-bed streams in the western United States. 128*. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Steffler, P., & Blackburn, J. (2002). *River 2D: Two-dimensional depth averaged model of river hydrodynamics and fish habitat. Introduction to depth averaged modeling and user's guide*. Edmonton, AB: University of Alberta.

Stewart, D. J., Kitchell, J. F., & Crowder, L. B. (1981). Forage fishes and their salmonid predators in Lake Michigan. *Transactions of the American Fisheries Society*, *110*, 751–763. https://doi.org/10.1577/1548-8659(1981)110<751:FFATSP>2.0.CO;2

Stratus Consulting Inc. (2010). Restoration plan and environmental assessment for the Upper Arkansas River watershed. Boulder, CO.

Thompson, D. M. (2002). Long-term effect of instream habitat-improvement structures on channel morphology along the Blackledge and Salmon Rivers, Connecticut, USA. *Environmental Management*, *29*, 250–265. https://doi.org/10.1007/s00267-001-0069-0

Waddle, T., & Steffler, P. (2002). R2D_Mesh: Mesh generation program for River2D two dimensional depth averaged finite element. Introduction to mesh generation and user's manual. US Geological Survey.

Wesche, T. A., Goertler, C. M., & Frye, C. B. (1987). Contribution of riparian vegetation to trout cover in small streams. *North American Journal of Fisheries Management*, *7*, 151–153. https://doi.org/10.1577/1548-8659(1987)7<151:CORVTT>2.0.CO;2

Wesche, T. A., Goertler, C. M., & Hubert, W. A. (1987). Modified habitat suitability index model for brown trout in southeastern Wyoming. *North American Journal of Fisheries Management*, *7*, 232–237. https://doi.org/10.1577/1548-8659(1987)7<232:MHSIMF>2.0.CO;2

Whiteway, S. L., Biron, P. M., Zimmermann, A., Venter, O., & Grant, J. W. A. (2010). Do in-stream restoration structures enhance salmonid abundance? A meta-analysis. *Canadian Journal of Fisheries and Aquatic Sciences*, *67*, 831–841. https://doi.org/10.1139/F10-021

Wohl, E., Angermeier, P. L., Bledsoe, B., Kondolf, G. M., Macdonnell, L., Merritt, D. M., … Tarboton, D. (2005). River restoration. *Water Resources Research*, *41*, 1–12.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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