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PHYSICAL AND BIOLOGICAL RESPONSES TO AN ALTERNATIVE REMOVAL STRATEGY OF A MODERATE-SIZED DAM IN WASHINGTON, USA

S. M. CLAESON[a*] AND B. COFFIN[b]

[a] *Pacific Northwest Research Station, USDA Forest Service, Wenatchee, WA, USA*
[b] *Mt. Adams Ranger District, USDA Forest Service, Trout Lake, WA, USA*

ABSTRACT

Dam removal is an increasingly practised river restoration technique, and ecological responses vary with watershed, dam and reservoir properties, and removal strategies. Moderate-sized dams, like Hemlock Dam (7.9 m tall and 56 m wide), are large enough that removal effects could be significant, but small enough that mitigation may be possible through a modified dam removal strategy. The removal of Hemlock Dam in Washington State, USA, was designed to limit channel erosion and improve fish passage and habitat by excavating stored fine sediment and reconstructing a channel in the former 6-ha reservoir. Prior to dam removal, summer daily water temperatures downstream from the dam increased and remained warm long into the night. Afterwards, a more natural diel temperature regime was restored, although daily maximum temperatures remained high. A short-lived turbidity pulse occurred soon after re-watering of the channel, but was otherwise similar to background levels. Substrate shifted from sand to gravel–cobble in the former reservoir and from boulder to gravel–cobble downstream of the dam. Initially, macroinvertebrate assemblage richness and abundance was low in the project area, but within 2 years, post-removal reaches upstream and downstream of the dam had diverse and abundant communities. The excavation of stored sediment and channel restoration as part of the dam removal strategy restored river continuity and improved benthic habitat while minimizing downstream sedimentation. This study provides a comparison of ecological effects with other dam removal strategies and can inform expectations of response time and magnitude. Published 2015. This article is a U.S. Government work and is in the public domain in the USA.

KEY WORDS: Hemlock Dam; restoration; temperature; turbidity; sediment; macroinvertebrate; dam removal

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INTRODUCTION

There are more than 76 000 dams in the USA, not including many small, run-of-the-river dams (Heinz, 2002). Most dams were installed prior to current-day environmental assessments and have significantly altered the ecology of rivers and watersheds. Dams and the stream habitat changes they bring about are implicated as a cause of biodiversity loss and are a known or suspected threat for native fish (Petts, 1984). As dams fill with sediment become unsafe or inefficient, or otherwise outlive their usefulness, dam removal is increasingly being considered for river restoration. Environmental responses to dam removals are likely to vary because of differences in the type of dam, quantity of sediment stored behind dams, and the hydrologic and geomorphic context of the river (Graf, 2003). Over many years, a river's ecosystem adjusts to the presence of a dam such that the dam's removal can cause a relatively large disturbance. The method of dam removal can also influence the magnitude of impact and the temporal and spatial scale of recovery of the associated aquatic ecosystem (O'Connor *et al*., 2015). Documenting the effects and recovery from removal of dams varying in size, type, and removal strategy is an important step in establishing benchmarks for future comparisons and determining the best method of dam removal given specific restoration objectives.

Dam removal will likely affect physical, chemical, and biological components of river ecosystems, and the responses of these different components may be intertwined (Stanley *et al*., 2002; Thomson *et al*., 2005; Maloney *et al*., 2008; O'Connor *et al*., 2015). Geomorphic responses are influenced by the quantity of sediment stored in the reservoir and the ability of the fluvial system to redistribute the sediment downstream (Doyle *et al*., 2005). Biological responses to dam removal are more influenced by small-scale habitat changes (Tullos *et al*., 2014). Therefore, the fate and management of trapped sediment in the reservoir is usually the greatest concern, even with removal of small dams (Downs *et al*., 2009). The release of sediment stored behind dams can degrade water quality, release contaminants, bury ecologically sensitive downstream habitats, aggrade the downstream channel, increase flood risks, and create unstable channels in the former reservoir (Graf, 2003). On the other hand, downstream benthic habitat may benefit from

*Correspondence to: S. M. Claeson, Pacific Northwest Research Station, USDA Forest Service, Wenatchee, WA, USA.
E-mail: sclaeson@fs.fed.us

receiving sediment that had been trapped in the reservoir if sediment levels downstream from the dam have been depleted over time (Schuman, 1995; Downs *et al*., 2009).

The goal of most removal projects is to minimize the negative effects of dam removal, while maximizing the rate of recovery of the physical and biological systems (Doyle *et al*., 2005). Hemlock Dam on Trout Creek, Washington, USA, impeded migration of an important run of USA-threatened Lower Columbia River steelhead (*Oncorhynchus mykiss*) and other migratory fish (Pacific Lamprey, *Entosphenus tridentate*, and introduced Chinook Salmon, *O. tschawytscha*). This moderate-sized dam was removed during the summer of 2009 at a cost of slightly less than $3 million. Because potentially high volumes of sediment released in late summer during dam removal were anticipated to harm juvenile steelhead and lead to habitat degradation, most of the reservoir sediment was excavated prior to dam removal. Dam removal operations began with diverting the stream flow around the reservoir and dam in a large pipe from 1 July to 14 August 2009. The reservoir was drained and approximately 42 000 m^3 of fine sediment was excavated, more than 4000 dump truck loads. Next, a channel was constructed through the former reservoir following the historic channel route. More than 2000 m^3 of gravel and cobble substrate were added to the sandy streambed of the new channel. Large logs and stumps were placed along the constructed channel banks and throughout the floodplain to help stabilize the new channel and the remaining sediment. A smaller side channel was constructed with an elevated streambed to alleviate main channel flow and erosion risk during high-flow events. Finally, the concrete dam and associated structures were removed over the course of a few days. The stream flow was returned to the new channel on 14 August 2009. Post-removal restoration activities included planting the former reservoir floodplain with native shrubs and trees, controlling invasive vegetation, and conducting stream habitat and aquatic biota monitoring.

This intensively managed strategy for the removal of Hemlock Dam and reservoir sediment was intended to address key factors limiting steelhead recovery in Trout Creek. Specific goals were to reduce peak water temperatures, increase aquatic habitat for fish, improve fish migration, and limit suspended sediment during dam removal operations to background levels. In this study, we monitored various stream metrics to determine if the dam removal goals, except fish migration, were met within three after removal. Specifically, we assessed water temperature and turbidity (as a surrogate for suspended sediment), channel substrate, and benthic macroinvertebrates before and after dam removal, and in comparison with an upstream reference reach. We expected the metrics to vary in their time frame of recovery (i.e. approximating the central tendency and mean of the reference reach). Water temperatures and turbidity were predicted to recover within a few days after dam removal. Channel substrate composition was predicted to change quickly with the augmentation of coarse substrates after dam removal compared with before conditions, but a longer time frame involving multiple high-flow events may be needed to redistribute substrates. Macroinvertebrates, an important component in stream food webs and nutrient cycling, were predicted to have the slowest rate of recovery and would respond as benthic habitat returned to a more natural lotic and stable condition.

STUDY AREA

Hemlock Dam spanned Trout Creek at river 2.5 km from its confluence with the Wind River in the Gifford Pinchot National Forest of southwestern Washington (Figure 1). Trout Creek drains 89 km^2 of managed coniferous forest where flow is dominated by runoff during rainfall or snowmelt. Mean annual discharge on Trout Creek is 7.1 $m^3 s^{-1}$, ranging from 0.3 $m^3 s^{-1}$ during late summer to more than 60 $m^3 s^{-1}$ during winter bankfull floods. Trout Creek comprises 10–20% of Wind River discharge (Figure 2; USDA Forest Service gauge #14128500). A wooden splash dam was built on Trout Creek in 1903 followed by a concrete dam in 1935. In 1951, the dam was modified for water storage and irrigation, along with the installation of a fish ladder for salmon and trout. The final dam was 7.9 m tall and 56 m wide. By 1997, the services that the dam provided were no longer needed. Instead the antiquated dam and fish ladder provided poor fish passage, disrupted sediment transport, inundated riverine habitat upstream, and increased water temperatures (Magirl *et al*., 2010; Buehrens *et al*., 2014). The 6-ha reservoir behind the dam had created a broad, shallow impoundment filled with up to 71 100 m^3 of sediment (Randle and Greimann, 2004). Reservoir sediment was predominantly sand with a median grain size of 0.6 mm and volume was $<$10% silt and $<$10% gravel and cobble. The channel downstream of the dam was largely scoured to bedrock and lacked gravel and cobble substrates.

METHODS

Sampling design

This study employed a before–after, control–impact design with a single control reach and one or more impact reaches depending on the response parameter. Monitoring activities were distributed among four stream reaches along Trout Creek spanning the areas above and below Hemlock Dam and Reservoir (Figure 1). The ‘upstream’ reach (500-m long) was located upstream of both the dam and reservoir in a continuously free-flowing stretch of the river and was intended to act as a reference reach. The upstream reach

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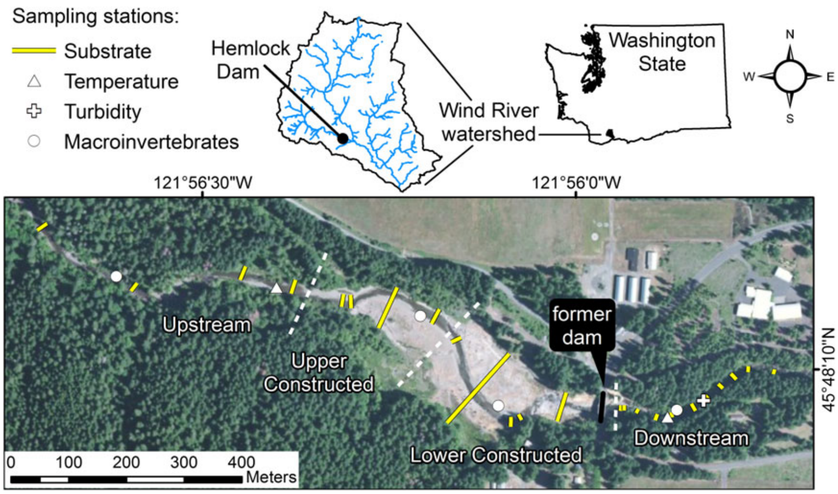

Figure 1. Hemlock Dam project area and the location of monitoring stations and transects relative to the former dam and reservoir. Hemlock Dam was located on Trout Creek, a major tributary to the Wind River in southwest Washington State. White dashed lines indicate study reach divisions. Not shown are the upstream turbidity station located 1.5 km upstream of the dam and two additional substrate cross sections located 1.5 km downstream of the dam. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

was unconfined and bordered by mature riparian trees and shrubs providing partial shade during the day. The 'downstream' reach (315-m long) was located immediately downstream of the dam in a more narrow and confined section of the river. The downstream reach had nearly complete canopy cover from mature riparian trees and steep hillsides. No restoration activities occurred downstream of the dam. Two stream reaches were established within the former reservoir, 'constructed upper' (320-m long) and 'constructed lower' (400-m long) (Figure 1). These reaches had no canopy cover as they were bordered by newly planted riparian vegetation. The constructed upper and lower reaches were monitored separately because they had different site-specific characteristics, but they both had gravel added as part of the channel construction. The upper reach had the potential to be more influenced by upstream non-dam-related inputs, was of low gradient, and located just downstream of a large, engineered log jam created during the restoration to improve fish habitat. The lower reach had a slightly steeper gradient, was more influenced by bedrock formations, and had more hill-shading and fewer in-stream wood structures.

Water temperature

Trout Creek water temperature (°C) was recorded hourly (HOBO Water Temperature Pro V2 Data Loggers) at one location 650 m upstream of the dam in the reference reach and one location 125 m downstream of Hemlock Dam (Figure 1). Temperatures were recorded from approximately July through September for 5 years prior to dam removal (2002,

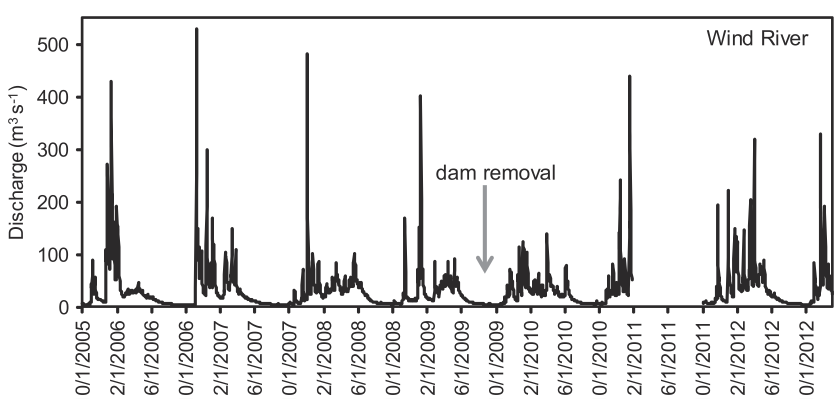

Figure 2. Mean daily discharge ($m^3 s^{-1}$) of the Wind River from 1 October 2005 to 31 December 2012. Trout Creek comprises 10–20% of Wind River discharge. Dam removal on Trout Creek occurred July–August 2009. Discharge was unavailable from 25 January to 30 September 2011

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2003, 2005, 2006, and 2008) and for 3 years after dam removal (2010, 2011, and 2012). Water temperature daily maxima, minima, and diurnal variation were calculated for both locations.

Turbidity

Turbidity is often used as a surrogate for suspended sediment and, while not perfect, can be a reasonable representation when the goal is to compare the magnitude and timing between river locations (Lewis, 1996). Turbidity (NTU) in Trout Creek was recorded hourly throughout the dam removal project in 2009 using two YSI 6136 Turbidity Sondes that were deployed at fixed locations to allow for comparison of upstream (background) and downstream (removal-influenced) turbidity levels. The upstream instrument was located approximately 1.5 km upstream of the dam, and the lower instrument was 150 m downstream of the dam (Figure 1). Each instrument was housed in a steel-well casing that was perforated and anchored to the channel bed. The instruments were in operation during the summer of 2009. On 14 August 2009, around 12:00, the downstream Sonde turbidity instrument was buried in sand resulting in an erroneous reading and then scoured clear within an hour by 13:00. In order to estimate the peak turbidity during that time period, we used linear interpolation over time to fit lines to the rising and falling slopes and determined the intersection value. Turbidity was also measured manually from the thalweg with Hach 2100P Portable Turbidimeters at two locations farther downstream of the dam, 1.5 km and 2.5 km downstream, during 14–15 August 2009 because of concern for high suspended sediments during the rewatering stage. Comparisons of turbidity values between Sondes and Hach instruments should be interpreted with caution because the instruments measure turbidity differently (e.g. portable turbidimeters tend to underestimate turbidity) (Anderson, 2005). Turbidity measures were not analysed statistically, but compared qualitatively between the downstream impacted sites and upstream reference site during dam removal operations in 2009.

Substrate

Benthic substrate compositions were measured at each of the four study reaches plus an additional reach farther downstream before (2007) and after dam removal (2009–2012), with the exception of the upstream and downstream reach soon after dam removal in 2009. Substrate counts were conducted using zigzag bankfull cross-channel transects with step length and boot toe particle selection (Bevenger and King, 1995). Four transects were located in the upstream reference reach, 4 transects in the constructed upper reach, 5 transects in the constructed lower reach, 11 transects in the downstream reach, and 2 transects located 1.5 km downstream of the dam (Figure 1). A minimum of 100 particles per transect were measured along their intermediate axis and tallied into Wentworth size classes: silt and clay (0–0.062), sand (>0.062–2 mm), gravel (>2–64 mm), cobble (>64–256 mm), boulder (>256–2048 mm), large boulder (>2048 mm), and bedrock.

Macroinvertebrates

Benthic macroinvertebrates were collected with a Surber sampler of mesh size 500 μm and sampling area of 1 ft^2 (0.09 m^2). Macroinvertebrates were collected in early October, while flows were low, before (2007) and after dam removal (2009–2011). Each year, six samples were randomly collected from flowing water habitat in a 50 m reach (Li *et al.*, 2001) within each of the upstream, constructed upper and lower, and downstream study reaches. However, no samples were collected from the downstream reach during the fall of 2009 due to restricted access, nor from the reservoir reaches prior to dam removal when the lake was present. A total of 25 529 aquatic macroinvertebrates were collected and identified to the lowest reliable taxonomic level (generally species or genus), except for Chironomidae, which were identified to subfamily, and non-insects to class. Individuals were assigned taxa traits of functional feeding group (FFG), voltinism, and tolerance to degraded habitat, such as disturbance, high temperatures, and fine sediments (Merritt *et al.*, 2008). Macroinvertebrate abundance per sample was standardized to the number of individuals per square metre.

Data analysis

To determine if summer water temperatures changed significantly after dam removal, upstream temperature metrics (daily maxima, minima, and diurnal variation) were regressed against downstream metrics for each day using generalized least squares (GLS) to account for serial autocorrelation in the residuals. The GLS modelling was performed in R (v.3.1.2) using the Linear and Non-linear Mixed Effects Models (NLME) package. The autocorrelation structure was modelled as an auto-regressive moving average with ‘p’ and ‘q’ chosen to minimize AIC. The models included treatment period (pre-treatment versus post-treatment) as the fixed effect of interest and treated year as a random effect. Regression diagnostics included time series analysis of residuals. As a separate analysis, the mean time of day that the daily maximum occurred at and the number of hours the temperature remained at or within 1 °C of maximum was calculated for each year and then averaged over the pre-removal years and post-removal years.

To test for significant changes in median substrate sizes over time, before and after dam removal, the size (mm) percentage of substrate less than D50 was calculated for each

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transect. Analysis of covariance (R v.3.1.2) was used to test for significant changes in the D50 (log transformed) for each Reach*Year interaction, with the upstream reference reach and year 1 (pre-removal year 2007) as the linear model intercept. A significant Year*Reach(X) term indicates that the values at Reach(X) over time have a slope (i.e. time trend) that is significantly different from the slope of the values at the upstream reach over time, and the coefficient indicates the direction and steepness of the slopes.

Generalized linear models (R v.3.1.2) were used to test for significant changes of *a priori* selected macroinvertebrate taxa groups over time for each reach. Results again focus on the Reach*Year interaction (as with the substrate analysis). Taxa groups analysed with Poisson error distributions included taxa richness; combined abundance of Ephemeroptera, Plecoptera, and Trichoptera (EPT); and Chironomidae abundance (Chi). Taxa groups with a binomial error distribution included the proportion of individuals that are clingers; multi-voltine; tolerant of habitat degradation; and within the FFG categories of collector-filters, collector-gathers, scrapers, shredders, or predators.

RESULTS

Water temperature

Prior to dam removal, summer water temperatures downstream of the dam were generally warmer and had less diurnal fluctuation than upstream of the reservoir (Figure 3). Post-removal daily maximum temperatures increased by an average 0.4 °C ($p = 0.0485$), daily minimum temperatures decreased by an average of 2.2 °C ($p < 0.0001$), and diurnal fluctuation increased by an average of 2.6 °C ($p < 0.0001$) compared with pre-removal. Dam removal resulted in a slight increase in daily maximum temperatures, but minimum temperatures that were nearly equal to those upstream of the dam (along the 1:1 line), resulting in water downstream of the former dam with greater diurnal fluctuation than upstream (Figure 3). The hourly water temperature pattern within a 24-h period also changed dramatically before and after dam removal. Before removal, the average time of day at which the maximum temperature occurred was 15:00 upstream and 19:00 downstream of the dam, a 4-h delay. In addition, water temperatures remained within 1 °C of the daily maximum temperature for 3.4 h longer downstream than upstream. After removal, the mean maximum temperature downstream was only delayed by 1 h compared with upstream (from 15:00 upstream to 16:00 downstream), and temperatures remained within 1 °C of daily maximum for approximately the same amount of time at both locations (<0.5 h difference).

Turbidity

Turbidity at the upstream location remained low, 3-5 NTU, during dam removal in the summer of 2009. During channel construction and actual dam removal, when most of the water was diverted around the reservoir and dam site (29 July to 13 August 2009), turbidity 150 m downstream of the dam did not exceed 13 NTU. However, at approximately 12:00 on 14 August, 2 h after re-watering the constructed channel (i.e. removal of the diversion pipe), single pulse of turbidity occurred and travelled downstream becoming more attenuated with distance. Downstream turbidity increased to an estimated peak of 670 NTU, but decreased to 16 NTU 24 h later (Figure 4). During this same time period the turbidity 1.5 km and 2.5 km downstream of the dam reached a peak of 240 NTU and 93 NTU, respectively. From 16 to 26 August, turbidity 150 m downstream of the dam remained low, ranging between 5 and 23 NTU.

Substrate

Benthic substrate at the upstream reach was dominated by gravel and cobble both before and after dam removal

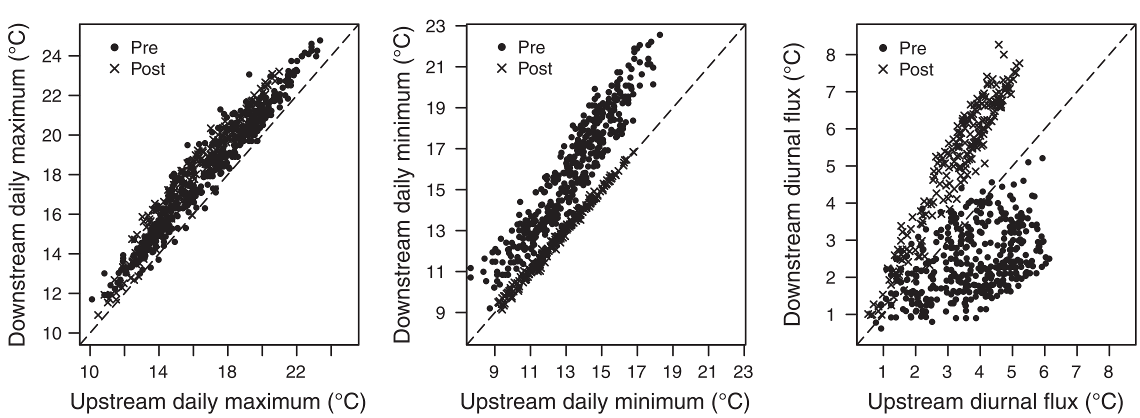

Figure 3. Trout Creek summer daily maximum, daily minimum, and diurnal fluctuations of water temperatures (°C) from sites upstream (x-axis) and downstream (y-axis) of the dam, before and after dam removal

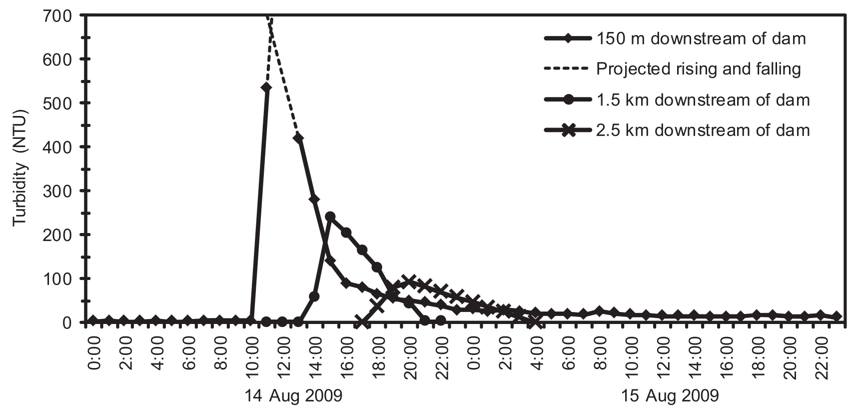

Figure 4. Turbidity (NTU) downstream of the former dam during the re-watering of Trout Creek on 14–15 August 2009. Sometime between 11:00 and 13:00 on 14 August, the YSI turbidity instrument 150 m downstream of the dam was buried and then scoured clean resulting in no data collection at 12:00. The projected rising and falling values during this time were predicted using linear interpolation over time. Upstream turbidity did not exceed four NTU on these dates (not shown on figure)

(Figure 5). The upper and lower constructed reaches were mostly sand while inundated by the reservoir before dam removal, but shifted to gravel and cobble dominated after dam removal from 2009 to 2012 with increasing median substrate size across years (Table I; $p = 0.0002$ and $p < 0.0001$, respectively). Prior to dam removal, the reaches within 300 m and 1.5 km downstream of the dam were dominated by boulders, but shifted to gravel and cobble after dam removal with decreasing median size across years ($p = 0.0669$ and $p = 0.0205$, respectively). However, the downstream reaches both before and after dam removal maintained 10–20% bedrock.

Macroinvertebrates

Overall, 74 aquatic macroinvertebrate taxa were recorded from Trout Creek between 2007 and 2011. Taxa richness of benthic assemblages in the upstream reference reach was higher and more consistent between years than in reaches disturbed by the dam removal. Upstream mean taxa richness ranged from 22 to 31 over the years surveyed with a low proportion of individuals tolerant to degraded habitat conditions, such as disturbance, high temperatures, and fine sediments (Figure 6). Abundances were dominated by collector-gathering insects (primarily *Ephemerella tibialis* mayflies and Chironominae midge larvae), followed by scrapers of periphyton (e.g. *Glossosoma* caddisflies, *Epeorus*, and *Cinygmula* mayflies).

Two months after dam removal in 2009, macroinvertebrate assemblage richness and abundances were relatively low in both of the constructed reaches in the former reservoir compared with the upstream reach (Figure 6). The upper and lower constructed reaches had a mean of 11 and 9 taxa identified, respectively, and were dominated by the

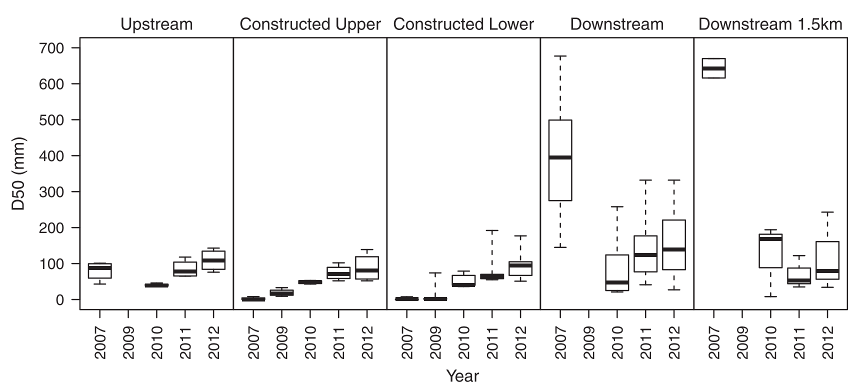

Figure 5. Substrate D50 (mm) from bankfull transects surveyed before (2007) and after (2009–2012) dam removal within each study reach: upstream (4 transects), constructed upper (4 transects), constructed lower (5 transects), downstream (11 transects), and downstream 1.5 km (2 transects). No data were collected at the upstream and downstream reaches soon after dam removal in 2009. Substrate types associated with grain size were silt and clay (0–0.062), sand (>0.062–2 mm), gravel (>2–64 mm), cobble (>64–256 mm), boulder (>256–2048 mm), and large boulder (>2048 mm). Plots show the median value (thick line), inner quartile range (box), and full range of values (dashed lines)

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Table I. Statistical summary results for substrate-D50 analysis of covariance linear model (114 residual d.f.) and for macroinvertebrate generalized linear models (70 residual d.f.)

Model	Response variable	Year*Reach coefficient (p-value)			
		Year*ConUp	Year*ConLow	Year*Down	Year*Down1.5 km
LM[a]	Substrate-D50	0.34 (0.0002)	0.41 (<0.0001)	−0.14 (0.0669)	−0.25 (0.0205)
GLM[b]	Taxa richness #	0.32 (0.0017)	0.30 (0.0054)	0.20 (0.0239)	NA
GLM[b]	EPT #	0.54 (0.0051)	0.17 (0.3802)	0.32 (0.0845)	NA
GLM[b]	Chironomidae #	0.68 (0.0460)	0.35 (0.3254)	0.27 (0.4340)	NA
GLM[c]	Collector-filter %	−0.34 (0.3826)	−0.33 (0.3544)	−1.01 (<0.0001)	NA
GLM[c]	Collector-gather %	0.25 (0.2993)	0.09 (0.7138)	0.16 (0.2601)	NA
GLM[c]	Scraper %	−0.16 (0.5692)	−0.06 (0.8421)	0.33 (0.0936)	NA
GLM[c]	Shredder %	0.18 (0.4908)	−0.01 (0.9911)	−0.36 (0.0014)	NA
GLM[c]	Predator %	−0.19 (0.2701)	−0.01 (0.9490)	0.20 (0.1576)	NA
GLM[c]	Multi-voltine %	0.35 (0.1638)	0.07 (0.7916)	−0.02 (0.9222)	NA
GLM[c]	Clinger %	−0.62 (0.1433)	−0.65 (0.1563)	−0.09 (0.6248)	NA
GLM[c]	Tolerance %	−0.32 (0.4090)	−1.13 (0.0004)	−0.19 (0.2012)	NA

LM, linear model; GLM, generalized linear models; d.f., degrees of freedom; EPT, Ephemeroptera, Plecoptera, and Trichoptera. All models included predictor variables of Year and Reach and their interaction. Year is a continuous variable where year 1 is 2007 (pre-removal), year 2 is 2009 (post-removal), and so on. Reach is a four-factor or five-factor variable representing the upstream, constructed upper (ConUp), constructed lower (ConLow), downstream (Down), and downstream by 1.5 km (Down1.5 km) reaches in Trout Creek. Macroinvertebrates were not collected from the far downstream reach. Results are presented for the Year*Reach interaction with the upstream reach and year 1 as the linear model intercept. Therefore, a significant Year*Reach(X) term indicates the time trend (i.e. slope) at Reach(X) is significantly different from the time trend at the upstream reach, and the coefficient indicates the direction (+/−) and steepness of the slopes.

[a]Response was log(x + 1)-transformed and fitted with a normal error distribution.

[b]Response was count data and fitted with a quasi–Poission error distribution and log-link.

[c]Response was proportional data and fitted with a quasi-binomial error distribution and logit-link.

mayflies *E. tibialis*, *Baetis tricaudatus*, and *Cinygmula*. Individuals tolerant to degraded habitats, primarily *B. tricaudatus* and *Paraleptophlebia bicornuta* mayflies, were high in abundance, especially in the lower reach (Figure 6). From 2009 to 2011, the upper and lower reaches increased approximately threefold in taxa richness (Table I; $p = 0.0017$ and $p = 0.0054$, respectively). The upper reach had a 10-fold increase in mean EPT abundances ($p = 0.0051$) and 50-fold increase in mean chironomid abundances ($p = 0.0460$) over time. The lower reach also increased in EPT and chironomid abundances but with greater variability over time ($p = 0.3802$ and $p = 0.3254$, respectively). The proportion of tolerant individuals decreased fourfold in the upper reach ($p = 0.4090$) and sevenfold in the lower reach ($p = 0.0004$). There were no significant changes to FFG or multi-voltine abundance proportions in these reaches compared with the upstream reference reach over time (Table I).

Downstream of the dam prior to removal, macroinvertebrate assemblages had a mean richness of 10 taxa and low EPT and chironomid abundances (Figure 6). On average, 30% of the individuals were tolerant to degraded habitats. Small-sized invertebrate shredders (*Zapada* stoneflies and *Micrasema* caddisflies) and collector-filterers (primarily *Simulium* black fly larvae) were proportionately more abundant downstream of the dam (35% and 23%, respectfully) than compared with upstream (7% and 3%, respectfully). On the other hand, scraping insects were proportionately low downstream (7%) compared with upstream (25%). Reservoirs tend to contain relatively high levels of autotrophic production, discharging sestonic organic matter to downstream consumers influencing FFG's in regulated rivers (Petts, 1984). After dam removal, mean taxa richness downstream increased nearly threefold from 2007 to 2011 (Table I; $p = 0.0239$). Mean EPT and chironomid abundances increased threefold and 12-fold ($p = 0.0845$ and $p = 0.4340$, respectively), but were slower to reach upstream levels than in the constructed reaches (Figure 6). The mean proportion of tolerant individuals decreased fivefold over time ($p = 0.2012$). By 2011, invertebrate shredders had decreased fourfold to 8% ($p < 0.0001$), collector-filterers decreased 11-fold to 2% ($p = 0.0014$), and scrapers increased fivefold to 34% ($p = 0.0936$), all more similar to upstream assemblages with 7% shredders, 7% filterers, and 30% scrapers.

DISCUSSION

The size of a dam and river limits the range of feasible dam removal and restoration strategies. Small dams are easier to manipulate, and their removal is often expected to have minor ecological impact. Removal of high-head dams on large rivers are more difficult to implement, and effects may be correspondingly large (Doyle *et al*., 2005; Downs *et al*.,

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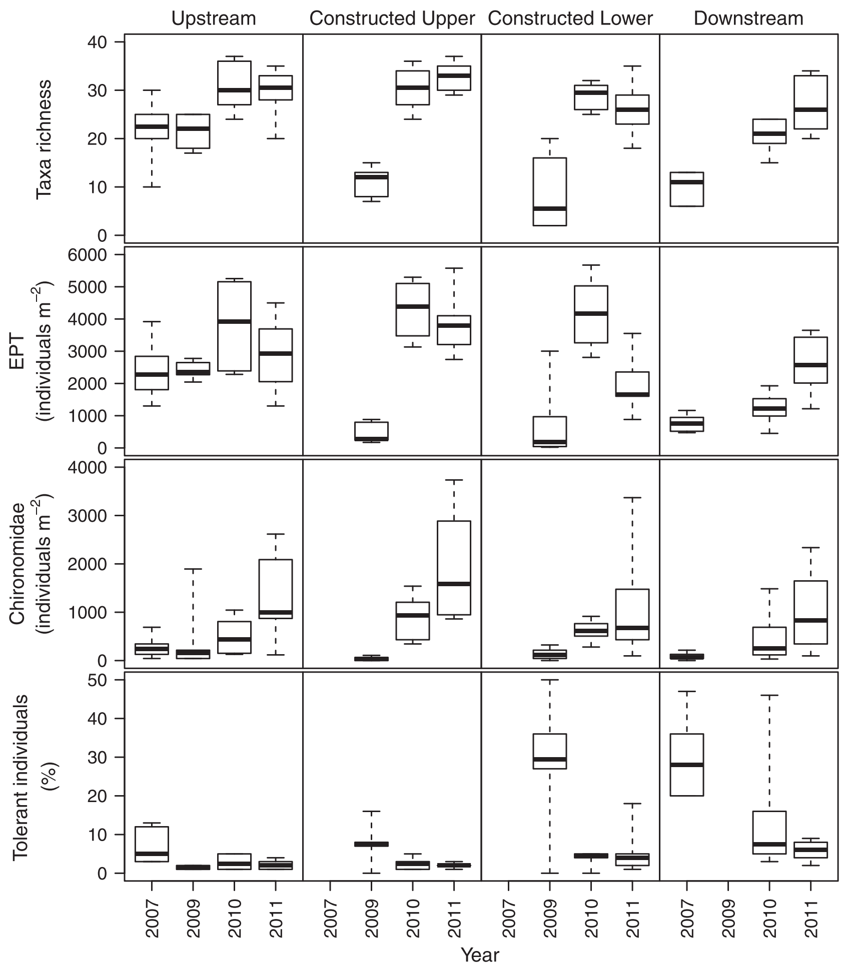

Figure 6. Macroinvertebrate groups of taxa richness, Ephemeroptera, Plecoptera, and Trichoptera (EPT) abundance, chironomidae abundance, and percent tolerant individuals collected before (2007) and after (2009–2011) dam removal within each study reach (six samples/reach/year). No macroinvertebrates were collected from the reservoir in 2007, as constructed reaches were created post-dam removal, and at the downstream reach soon after dam removal in 2009. Plots show the median value (thick line), inner quartile range (box), and full range of values (dashed lines)

2009). Moderate-sized dams, like Hemlock Dam, are large enough that removal effects could be significant, but small enough that mitigation may be possible through a modified dam removal strategy (Magirl *et al*., 2010). This study provides a point of comparison with other dam removal projects in which less aggressive measures have been taken to control sediment erosion, establish cobble-dominated streambeds, and improve aquatic habitat for fish.

Water temperature

When the dam was present, the thermal inertia in the reservoir kept the water at a relatively constant temperature such that the downstream water remained warm throughout the night. Continuously warm water flowing over the dam may have suppressed downstream abundance and richness of macroinvertebrate taxa that require cold water thereby reducing food resources for fish (Benjamin *et al*., 2013). These high water temperatures and altered diel cycles were also found to affect the growth performance of juvenile steelhead downstream of Hemlock Dam (Sauter and Connolly, 2010). Bioenergetic studies of fish indicate a strong positive relationship between feeding rates and metabolism with temperature provided food is available (Wootton, 1998). Juvenile salmon and trout primarily feed at dawn and dusk when benthic insects are most active, but when high temperatures extend into the evening, fish may have to forage in thermally stressful conditions (Quinn, 2005). Temperature and food also have a strong effect on life history diversity and age at smolting for steelhead (Benjamin *et al*., 2013).

To improve conditions for cold water fish, one of the goals of the dam removal was to reduce water temperatures in Trout Creek during low-flow summer months. Although

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summer daily maximum temperatures were not reduced, minimum temperatures were reduced, and the 24-h temperature regime returned to a normal lotic pattern. Maximum temperatures likely remained high after dam removal because the channel through the former reservoir was open to sunlight allowing solar radiation to heat the water, as opposed to the riparian forested reach upstream. After dam removal, the riparian area in the former reservoir was replanted with vegetation such that stream temperatures may cool as tree shading develops. Riparian vegetation usually requires the greatest time for recovery after dam removal and can influence water temperature through stream shading (or lack thereof), sedimentation through bank stability, and macroinvertebrates through allochthonous inputs (Doyle *et al*., 2005).

Aquatic habitat

Another goal of dam removal restoration was to increase aquatic habitat beneficial for fish. Although fish responses are not addressed in this study, preliminary results by Buehrens and others (2014) indicate increased steelhead abundance in Trout Creek within a few years after dam removal. The quality and quantity of aquatic habitat for fish increased as measured by the return of a natural daily temperature regime, gravel and cobble substrates, and benthic macroinvertebrate assemblages.

Gravel and cobble substrates are preferred by native fishes for adult spawning and rearing habitat (Quinn, 2005). Substrate coarsened above the dam immediately after fine substrate was excavated, and gravel and cobble were deposited during channel reconstruction. Downstream of the former dam, the bedrock-dominated streambed became finer within 1 year after dam removal as gravel and cobble were transported from upstream and deposited downstream. Large flow events after dam removal (Figure 2) likely helped to redistribute gravel and cobble downstream, but did not cause major erosion of the reinforced stream banks in the former reservoir. During the large flow event of January 2011, the water crested the stream banks and flowed into the restored floodplain as anticipated (B. Coffin unpublished observations). Changes in substrate above and below the dam were beneficial to targeted macroinvertebrates and fish.

Benthic macroinvertebrate communities improved considerably 1 year later in the former reservoir and 2 years later downstream of the dam. We attribute these delays to colonization lag, unstable substrate, and low substrate heterogeneity. Although we did not measure substrate stability, we observed ankle-deep loose gravel and cobble while collecting benthic samples in the fall of 2009. Similarly, stream restoration involving sediment excavation and gravel augmentation in the Merced River, California, corresponded with reductions in benthic macroinvertebrate abundance and biomass that were driven by increased substrate mobility and decreased substrate heterogeneity (Albertson *et al*., 2011).

Benthic macroinvertebrate assemblages in other formerly impounded reaches have been reported to resemble those from continuously free-flowing reaches within 1 to 2 years, irrespective of channel restoration activities associated with dam removal (Stanley *et al*., 2002; Casper *et al*., 2006; Maloney *et al*., 2008). In our study, macroinvertebrate abundance, especially chironomids, was high at the upper constructed reach, even greater than upstream, possibly because of increased primary productivity from more sunlight exposure once the substrate stabilized. For example, a bottom-up effect driven by increased light was observed to promote biofilm production and Chironomidae abundance following logging along headwater streams (Kiffney *et al*., 2003). Chironomidae can respond quickly to increases in food resources and is relatively tolerant of disturbance. Thus, benthic communities may appear to recover relatively quickly, but until the impounded sediment has been removed, stabilized, or eroded, high rates of sediment transport during flood events may continue to disturb the benthos.

Turbidity

An important goal involving the dam removal strategy was to keep suspended sediment levels in Trout Creek at background levels in hopes of avoiding direct and indirect harm to downstream fish or habitat. This strategy resulted in a single pulse of turbidity of short duration when water was returned to the stream channel, but otherwise, turbidity remained low and similar to the upstream reference reach. Dam removals involving natural river erosion or the rapid release of impounded sediment can have negative effects on downstream benthic habitats and biota from sediment transport (Thomson *et al*., 2005; Orr *et al*., 2008). By contrast, dam removals involving sediment excavation had no significant downstream sediment aggradation (Wildman and MacBroom, 2005; Plante *et al*., 2009). Thus, potential negative effects on downstream fish and habitat after removal of Hemlock Dam were apparently mitigated by removing much of the sediment behind the dam.

CONCLUSION

The intensively managed removal strategy employed on Hemlock Dam resulted in relatively short-lived negative effects, and within 2 years, the stream ecosystem contained productive habitat for macroinvertebrates and attributes important for native fishes. Following dam removal, water temperatures and turbidity responded rapidly, whereas benthic substrate and macroinvertebrates required 1 to 2 years to recover. In this case, we believe that had reservoir sediments not been excavated nor coarse substrate added, high

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deposition loads of fine sediment and a lack of suitable substrate would have slowed recovery of benthic habitat and macroinvertebrate assemblages. As implemented, this dam removal strategy resulted in no major erosion events occurring within 3 years and improved physical and biological habitat components important to native fish.

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