A YEAR-LONG BIOPHYSICAL ASSESSMENT OF A PARTIALLY BREACHED WEIR DAM REMOVAL IN A SOUTHERN APPALACHIAN MOUNTAIN STREAM

A Thesis
by
MADISON BLUE SUTTMAN

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Abstract

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While dam removal is increasingly viewed as a viable approach to river restoration, a lack of comprehensive research hinders our ability to predict ecosystem responses. This study investigates the impact on and recovery of macroinvertebrate communities following removal of the Payne Branch Dam (6.7m high, 53.3m wide) and restoration of the former impoundment in the Middle Fork of the South Fork of the New River (Watauga County, NC). My objectives were to quantify the impacts on macroinvertebrate community structure, monitor the downstream transport of bed sediment, and assess the rate of recovery. I hypothesized that due to partial breach, dam removal mitigation efforts, and its status as a mid-sized dam in a third-order, medium-high gradient stream, the impact on macroinvertebrates would be minor, and downstream sites would resemble reference sites within 6-months post-removal.

Benthic macroinvertebrates were collected at eight sites, with four above and four below the dam, to calculate metrics of abundance, richness, community composition, diversity, and biotic index. Bed particle size measurements at each site addressed potential impacts of bed sediment transport on community assemblages.

Results revealed a fining in downstream grain size 3-months post-removal with recovery by 6-months. Marked reductions in downstream macroinvertebrate abundance and EPT richness were observed 3-months post-removal and minor differences between reference and impacted sites reemerged 12-months post-removal. Bed sediment sizes did not reliably predict macroinvertebrate community metrics in linear regression. Macroinvertebrate diversity was unaffected. Non-Metric Multidimensional Scaling (NMDS) plots identified an upstream outlier, but otherwise showed little variation in community similarity among sites of reference communities and impacted communities. The downstream North Carolina Biotic Index (NCBI) differed little from reference communities.

These findings suggest that dam removal-induced disturbances are temporary, with modest changes and rapid recovery observed in the study system. Ongoing differences in reference and downstream abundance suggest the presence of additional factors limiting downstream recovery. The study supports dam removal as a viable river restoration method but emphasizes the need for careful planning and assessment, with consideration of technique to adequately address the characteristics of each river and dam.

The Payne Branch Dam removal provided a unique opportunity to study the impact and response of benthic macroinvertebrate communities in a third-order, cold, mountain stream. As dam removals gain momentum, this study contributes to the growing body of knowledge, aiding future decision-making for streams of similar size and characteristics.

Keywords: macroinvertebrate, disturbance, river restoration, bed sediment, biotic index

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Dedication

This thesis is dedicated to my parents, for fostering my interests in the natural world and for always believing in me, regardless of where my ambitions take me.

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Foreword

This thesis will be submitted to the journal *Freshwater Biology* and has been formatted according to the style guide for this journal for minimal revisions before publication.

1 INTRODUCTION

Humans have been constructing dams for thousands of years, harnessing water flow for navigation, recreation, water storage and regulation, and hydroelectric power generation (Bellmore et al., 2017; Nilsson et al., 2005). However, the construction of dams alters a river's natural flow and sediment regimes, imposing ecological disturbances such as habitat fragmentation, restricted movement patterns, and changes in water temperature and chemistry (Bednarek, 2001; Poff et al., 1997). Additionally, 80% of dams have surpassed their working lifespan, rendering them impractical, unsafe, or requiring costly upkeep (Bellmore et al., 2017; Stanley and Doyle, 2003). Through this combination of aging infrastructure and the potential for river restoration, interest in dam removals has soared, with more than 1,500 documented dam removals having been completed in the United States this century (American Rivers, 2023). Yet, fewer than 10% of these projects have been accompanied by scientific research, limiting understanding of dam removal impacts and recovery (Bellmore et al., 2017). Studies lack duration, pre-removal monitoring, and geographic variation (Bellmore et al., 2017; Foley et al., 2017; Graf, 2005). Coupled with natural variation in river hydraulics and sediment regimes, limited research inhibits our ability to predict ecosystem response following dam removal.

1.1 Study species

Aquatic macroinvertebrates are commonly utilized as bioindicators of ecosystem health due to their sensitivity to changes in water quality (Merritt et al., 2019). This trait, along with their benthic nature, short generation times, diversity, and abundance make them ideal model organisms (Hart et al., 2002; Sullivan et al., 2018). Among them, the insect orders of Ephemeroptera, Plecoptera, and Trichoptera (EPT) are particularly susceptible to pollution, earning them the title of indicator taxa because their presence or absence in a system indicates its health (Merritt et al., 2019).

1.2 Dam impacts

The importance of stream connectivity has been well established with ecological paradigms such as the Natural Flow Regime (Poff et al., 1997), River Continuum Concept (Vannote et al., 1980), Nutrient Spiraling Concept (Ensign and Doyle, 2006), and Serial Discontinuity Concept (Ward and Stanford, 1983). In short, impoundments alter connectivity causing unnatural changes in streamflow, sediment dynamics, nutrient cycling, and thermal regimes that limit the distribution and abundance of biota (Lake, 2003). Impoundments limit the upstream travel of fish and other organisms necessary to find optimal habitat, water levels, and food availability (Bednarek, 2001; Kanehl et al., 1997). The trapping of sediment and organic material behind a dam causes downstream shifts in community composition, particularly for herbivores and detritivores that directly rely on organic material (Bednarek, 2001; Stanley and Doyle, 2003). Reduced taxa abundance and richness, changes in taxonomic diversity and composition, and a shift toward pollution-tolerant and lentic species have been reported as a result of impoundment (Bednarek, 2001; Gillette et al., 2005; Pollard and Reed, 2004). Furthermore, a change in community composition at one level of food chain can significantly affect organisms at higher trophic levels, creating a trophic cascade.

The presence of dams in river ecosystems creates a press disturbance, where biota are continually disturbed over long periods (Tullos et al., 2014). When dams are removed, it shifts to a pulse disturbance where the event is often significant, but recovery is observed in the weeks to months following (Foley et al., 2017; Tullos et al., 2014). Stream response to disturbance is primarily characterized by geomorphic factors such as channel adjustments, slope adjustments, and altered flow regimes, which are known to vary geographically, but other key factors include dam condition, removal methods, river size, location within the watershed, and sediment dynamics (Bellmore et al., 2017; Foley et al., 2017). Furthermore, effects can be exacerbated by the magnitude, frequency, duration, timing, and rate of change of disturbance (Poff et al., 1997).

Following dam removal, the previously lentic habitat suddenly shifts to a lotic one, and the water carries decades of accumulated sediment with it. Once sediment transport is initiated, it is hard to stop, and large quantities of sediment are rapidly suspended, eroded, and redistributed (Foley et al., 2017, Graf, 2005). This can cause the filling of downstream pools, widening of channels, and deposition of fines, resulting in changes in temperature and habitat availability (Bednarek, 2001; Foley et al., 2017; Tullos et al., 2016). Across taxa, decreased density, richness, and diversity have been observed (Kanehl et al., 1997; Orr et al., 2008; Pollard and Reed, 2004). Sediment accumulation and scouring resulting from dam removal can cause increased macroinvertebrate drift, burying of habitat, and the abrasion and clogging of respiratory organs (Bilotta and Brazier, 2008; Hart et al. 2002). Additionally, high nutrient input from previously trapped organic material can result in eutrophication events, causing changes in species composition (Folegot et al., 2021; Sullivan and Manning, 2017).

1.3 Removal and recovery

The removal of dams restores flow and reinstates connectivity, causing rivers to undergo geomorphological changes that may better align with a natural flow regime, supporting the necessary conditions for native flora and fauna. Recovery timelines for biological, physical, and geomorphic changes range from days and weeks to months, years, or even decades (Bushaw-Newton et al., 2002). Biota with rapid generation times may recover within days or weeks, whereas slow-developing organisms may take months or years or never fully recover (Bushaw-Newton et al., 2002; Poulos et al., 2014). Despite disparities in recovery time, the literature indicates that the ecological disturbances brought on by dam removals are temporary, and vegetation, invertebrate, and fish populations recover relatively quickly. With time, fish and macroinvertebrate diversity metrics become nearly indistinguishable from reference reaches (Doyle et al. 2005). Migratory fish can relocate upstream as needed to reach spawning habitats (Doyle et al. 2005). Recolonization of native riverine fish species occurs at the impoundment area, replacing the non-natives (Bednarek 2001). Similarly, a shift in macroinvertebrate communities from lentic to lotic specialists is observed (Hart et al. 2002). Colonizing plants can improve bank stabilization (Doyle et al. 2005).

Thus far, dam removal studies have focused on medium-sized dams even though the majority of dam removals are on small run-of-the-river dams (<2m height), while large dams (>10m height) are known to have the greatest impact on aquatic ecosystems (Bellmore et al., 2017; Foley et al., 2017). There is also a geographic disconnect with most study areas located in the Midwest followed by the Atlantic and West coasts (Bellmore et al. 2017). Without diverse local and regional studies, the ability of stakeholders to make accurate predictions regarding stream response is undermined. In addition, research has primarily focused on warm, low-gradient waterways (Gillette et al., 2016) and large rivers (Gangloff, 2013). Many studies have employed large sampling intervals on the scale of months to years, potentially overlooking a critical window of change in the weeks to months post-removal (Pollard and Reed, 2003; Mahan et al., 2021; Sullivan and Manning, 2017). Moreover, there has been little research on partially breached dams, but evidence suggests that a breach could exacerbate downstream scour, resulting in reduced density, diversity, and richness (Gangloff, 2013; Maloney et al., 2008).

1.4 Objectives

The objectives of this study were to quantify the impacts of dam removal on ecosystem health via changes in macroinvertebrate biotic indices and community assemblages, to identify the extent to which bed sediment transport affects these changes, and to measure the rate of recovery. Review of historical discharge data identified five discharge events in the year post-removal corresponding to depth at flood stage. Decreased abundance, diversity, and changes in the North Carolina Biotic Index (NCBI) were expected in downstream sites post-removal. A decrease in sensitive taxa such as Ephemeroptera, Plecoptera, and Trichoptera (EPT) was also expected. A downstream decrease in sediment grain size was expected. Downstream impacted communities were predicted to reflect upstream reference communities within the first 6-months post-removal, indicating recovery. The results of this study are intended to inform future decision-making processes regarding dam removals and may be particularly applicable to low-order, cold, mountain streams.

2 METHODS

2.1 Study area

The Payne Branch Dam (6.7m high, 53.3m wide) was situated on the Middle Fork of the South Fork of the New River (Watauga County, NC). The Middle Fork spans nine miles and is classified as third-order at the dam site. The sample reach ranges in elevation from 940-1,048m and has a medium-high gradient (Table 1). US-321, a four-lane highway, runs along the study reach from sites MF1-MF6. Land cover estimates of the study reach suggest it is 87.52% forested and 12.48% impervious surfaces (Tuberty and Colby, 2019). Pre-removal water chemistry parameters were measured using a YSI Quattro™ multimeter probe (YSI Inc, Yellow Springs, OH).

The Payne Branch Dam, a low-head weir dam constructed in 1924, was owned and operated by New River Light and Power for the production of electricity. At the time of removal, it had been decommissioned for nearly 50 years and was partially breached. The removal process was gradual, beginning in late July 2020 (0-months post-removal), and was more than 95% complete by November 2020 (1-month post-removal). In addition to the removal of the impoundment, restoration measures were implemented on a 1,200-linear-foot section in the former impoundment, which included the removal of 20,000 tons of excess sediment, grading of the banks and channel, and the planting of native trees and shrubs for bed stabilization. At the time of removal, there were two additional NCDOT bridge replacement construction projects on the Middle Fork, one located upstream of site MF2 and the other upstream of site MF3 (Fig. 1).

Eight sampling sites were selected; four upstream of the impoundment to serve as reference sites (MF1-MF4) and four downstream to serve as impaired sites (MF5-MF8) (Fig.

1). Sites were selected based on accessibility and distance from the impoundment. Upstream of site MF8, the Middle Fork converges with the East Fork and Winklers Creek to form the South Fork New River.

2.2 Discharge

U.S. Geological Survey historical discharge data for the South Fork of the New River near Jefferson, NC, located roughly 70 river kilometers downstream of the study area, was utilized to identify flow events likely to exceed bankfull discharge, indicating the potential to cause significant geomorphic change (USGS Water Data Services, 2023). Discharge events greater than 3,128cfs were selected by using a stage-discharge rating curve to calculate average discharge at flood stage depth (12ft) (National Weather Service, 2020).

2.3 Sediment monitoring

Pebble counts were conducted following a modified Wolman procedure (Wolman, 1954) where 100 particles were randomly selected from a 100-meter transect using the zig-zag method across the bankfull width (Bevenger and King, 1995). To reduce sampling and measurement variability, the particle located at the tip of my right toe was selected using the tip of my index finger, and the intermediate axis of the particle was measured utilizing a gravelometer (Cole-Parmer, Vernon Hills, IL) as a template. Particles were sorted according to the following size classes based on the smallest gap in which the particle could pass: 2mm, 2.8mm, 4mm, 5.6mm, 8mm, 11mm, 16mm, 22.6mm, 32mm, 45mm, 64mm, 90mm, 128mm, 180mm, and >180mm. From these measurements D16, D50, and D84 grain sizes were calculated for each site. D16 is the grain size at which 16% of particles are finer, representing fine grains. D50 is the median grain size or the size at which 50% of particles are finer, representing medium-sized grains. D84 is the grain size at which 84% of particles are finer, representing relatively coarse grains. Pebble counts were performed during removal at 0-months (September 2020) and post-removal at 3-months (January/February 2021), 6-months (May 2021), 9-months (August 2021), and 12-months (November 2021).

2.4 Benthic macroinvertebrate sampling

The first collection was conducted during removal at 0-months (July/September 2020), followed by subsequent post-removal collections at 1-month (November 2020), 2-months (December 2020/January 2021), 3-months (January/February 2021), 4-months (March 2021), 5-months (April 2021), 6-months (May 2021), 9-months (August 2021), and 12-months (November 2021). Benthic macroinvertebrates were sampled using a modified Qual 4 (NC Department of Environmental Quality, 2016) protocol as described below, to increase chances of detecting cryptic species and to standardize the collection duration. At each site, one riffle-kick was conducted by disturbing a 3.0m^2 area upstream of a 1.0m^2, 500µm kick net (BioQuip Products, Rancho Dominguez, CA). Organisms were picked from the net for fifteen minutes or until none remained. A 500µm D-net (BioQuip Products, Rancho Dominguez, CA) was utilized to conduct three sweeps from root mats or undercut banks. Macroinvertebrates associated with leaves were collected by gathering three separate handfuls of submerged leaves, rinsing them in a collecting pan, and picking over each leaf. When present, up to three sand samples were taken using an 8-inch #10 brass sieve (Cole-Parmer, Vernon Hills, IL) to wash out fine particles and retain specimens. Finally, a 15-

minute visual search was conducted to sample submerged woody debris, the undersides of rocks, and any additional habitats not previously sampled.

Specimens collected from each sampling technique were pooled and preserved in 80% ethanol in the field. Macroinvertebrates were identified to family according to the taxonomic keys in Merritt et al. (2008) and transferred to archival grade vials (Discount Vials, Madison, WI) in 80% ethanol.

Macroinvertebrate data was utilized to calculate metrics of abundance, richness, community composition, diversity, and biotic integrity. Total abundance refers to the total number of organisms at each site, while EPT abundance is the number of organisms at each site belonging to the orders of Ephemeroptera, Trichoptera, and Plecoptera. Family richness is the number of families recorded at each site, and EPT richness is the number of families belonging to the orders of Ephemeroptera, Trichoptera, and Plecoptera. The percent of organisms belonging to combined EPT orders, as well as those belonging to the orders of Ephemeroptera, Plecoptera, Trichoptera, Diptera, and other taxa were calculated. The Shannon-Wiener Diversity Index was adapted using the equation $H' = -\Sigma p_i * ln(p_i)$, where p_i is the proportion of the community made up of the ith family.

Family-level pollution tolerance values were assigned by averaging the genus-level tolerance values for all genera in a family as presented in the Standard Operating Procedure (NC Department of Environmental Quality, 2016). Family-level tolerance values were used to calculate North Carolina Biotic Integrity (NCBI) using the equation $B = (\Sigma(T_i)(n_i))/N$ where T_i is the tolerance value for the ith taxon, n_i is the relative abundance category (1, 3, or 10) for the ith taxon, and N is the sum of all abundance category values (NC Department of Environmental Quality, 2016).

2.5 Data analysis

Site MF1 was determined during data analysis to be an outlier due to its downstream proximity to the Blowing Rock wastewater treatment plant and Lake Chetola impoundment and was therefore excluded from any "reference" metrics for macroinvertebrates, although it was retained for sediment metrics.

Statistical analyses and data visualizations were conducted with RStudio® (version 2023.03.0+386, R Core Team, Auckland, New Zealand). Levene's test was used to assess whether the upstream reference population and downstream impacted population had equal variance for metrics of abundance, richness, community composition, diversity, and grain size. When the assumption of equal variance was met, Two-Sample T-tests were performed to determine whether there was a significant difference in the means of the upstream and downstream populations. In instances where the populations did not meet the assumption of equal variance, the Mann-Whitney U Test was performed. The statistical significance level (α) for this study was set at 0.05. Line graphs were created to visualize changes in metrics across time. Linear regression analysis with one-way ANOVA models were created to investigate the relationship of grain size on macroinvertebrate abundance and richness.

Non-Metric Multidimensional Scaling (NMDS) ordination plots were created using the vegan package from a family abundance matrix to view Morisita-Horn dissimilarity across sites, which was chosen due its robustness to differences in richness and highly abundant taxa. To assess community similarity of the upstream sites and downstream sites, a Permutational Multivariate Analysis of Variance (PERMANOVA) test was performed with the community composition data obtained through NMDS as the response variable and site as the explanatory variable.

3 RESULTS

3.1 Water chemistry

Temperature, dissolved oxygen, specific conductivity, calcium, total dissolved solids, and pH were consistent with historical data for the location (Table 2). Elevated specific conductivity and total dissolved solids at Site MF1 are attributed to runoff, road salts, and the Blowing Rock sewage effluents located upstream of the study area.

3.2 Discharge

Throughout the study period, there were five events greater than 3,128cfs (Fig. 2). Of those, one event occurred prior to the 0-month bed sediment and macroinvertebrate collection in September 2020. The next two events took place prior to the 1-month macroinvertebrate sampling period and 3-month bed sediment collection in November 2020. After that, the next event occurred during the 5-month macroinvertebrate collection and prior to the 6-month bed sediment collection in April 2021. The final discharge event occurred during the 9-month macroinvertebrate and bed sediment collection in August 2021.

3.3 Sediment

The D16 and D50 (Fig. 3) at impacted sites decreased post-removal, while they increased at reference sites, leading to a significant difference in D16 ($p=0.0200$) and D50 ($p=0.0440$) between reference and impacted sites. D16 and D50 subsequently increased at the next collection 6-months post-removal. No significant differences in D84 (Fig. 3) were detected among reference and impacted sites throughout the study period, although a large drop in grain size was observed 9-months post-removal.

3.4 Macroinvertebrates

3.4.1 Abundance

A total of 22,244 organisms and 64 families were collected. EPT abundance appears to reflect total abundance (Fig. 4), with EPT taxa making up between 79% and 91% of all downstream organisms and between 82% and 94% of all upstream taxa. EPT and total macroinvertebrate abundance decreased in the first month post-removal, before returning to pre-removal numbers in the subsequent sampling period. At 3-months post-removal, EPT and total abundance nearly doubled at upstream reference sites. There was no change downstream, but the difference in EPT abundance ($p=0.0109$) and total abundance ($p=0.0177$) downstream was significantly different from reference sites. At 5-months post-removal, there was a drop in EPT and total abundance at both reference and impacted sites before increasing and remaining steady throughout the remainder of the study. The differences between reference and downstream sites in EPT abundance ($p=0.0324$) and total abundance ($p=0.0152$) at 12-months post-removal was also significant.

3.4.2 Richness

EPT family richness (Fig. 5) drove family richness (Fig. 5) in a manner similar to abundance. Both EPT and total family richness showed differences between reference and impacted sites in pre-removal collections, but the observed differences were only significant for EPT family richness ($p=0.0393$). A decrease in upstream richness and an increase in downstream richness at 1-month post-removal eliminated the gap between reference and impacted sites, until 4-months post-removal where there was an increase in richness upstream but no change downstream. Again, this difference between reference and impacted sites is significant only for EPT family richness (p=0.0078). Upstream EPT family richness and total family richness decreased 5-months post-removal, simultaneously with an observed algal bloom, before rising again and remaining steady through the end of the study.

3.4.3 EPT Composition

There were no significant differences in percent EPT (Fig. 6) at reference and impacted sites. Percent Plecoptera (Fig. 7) was similar across reference and impacted sites with no significant differences. Ephemeroptera made up the majority of EPT taxa collected at all sites 3-months post-removal (Fig. 7). There was a significant difference in percent Trichoptera (Fig. 7) at 6-months post-removal (p=0.0034) with a greater percent Trichoptera at upstream sites than downstream ones. Conversely, there was a significant difference in percent Ephemeroptera 6-months post-removal (p=0.0145), which was greater at downstream sites than upstream ones. Percent Diptera (Fig. 7) increased in months 1-, 2-, and 4- post-removal, before decreasing at 5-months post-removal and exhibiting another gradual increase at 9- and 12-months post-removal. The difference at reference and impacted was significant at 1-month post removal (p=0.0341), with impacted sites having a greater percent Diptera.

3.4.4 Diversity

Shannon diversity (Fig. 8) exhibited a slight increase in diversity 1-month post-removal, before a drastic drop 2- and 3-months post-removal. By 5-months post-removal, the score returns to pre-removal levels, where it stayed until the end of the study. No significant differences in Shannon diversity among upstream and downstream sites were observed.

3.4.5 Community similarity

NMDS ordination of macroinvertebrate community data at both upstream (Fig. 9; stress=0.1401706) and downstream (Fig. 10; stress=0.1373864) sites resulted in stress values less than 0.15, indicating that both plots have fair goodness of fit. NMDS plots and PERMANOVA of upstream sites indicated that MF1 had poor macroinvertebrate similarity as compared to the other reference sites (p=0.006), identifying it as an outlier. With site MF1 excluded, PERMANOVA indicated no differences among sites (p=0.690) and NMDS stress improved (Fig. 11; stress=0.1109522). As a result, site MF1 was excluded from further macroinvertebrate data analysis but retained for sediment analysis. Downstream sites had substantial overlap in NMDS plots and PERMANOVA indicated that all downstream sites could be grouped together for further analysis (p=0.895).

3.4.6 North Carolina Biotic Index

NCBI bioclassification scores (Table 3; Fig. 12) showed similar water quality prior-to and in the first 5-months post-removal at upstream and downstream sites, with the majority of sites earning a bioclassification of "Good." At 6-months post-removal, upstream sites earned one "Excellent" classification. At 9-months, both upstream and downstream sites earned one "Excellent." At 12-months post-removal, all upstream sites earned a bioclassification of "Good," while three of four downstream sites earned a bioclassification of "Excellent," showing potentially a slight improvement in downstream water quality 9- and 12-months post-removal.

3.5 Linear regression

Linear regression analysis indicated no significant relationship between macroinvertebrate abundance and richness metrics and grain size metrics (Fig. 13 & Fig. 14). Additionally, models could only explain a small fraction of the variance (<5%), as characterized by low R-squared values.

4 DISCUSSION

The primary goal in dam removal projects is to limit negative effects, while maximizing the rate of recovery (Doyle et al., 2005). Research suggests that mid-sized dams have the potential for significant undesired effects but remain small enough to mitigate such effects with restoration strategies (Claeson and Coffin, 2016). Dam removal approaches play a crucial role mitigation, yet few studies have reported the approach utilized. Instant removal involves the rapid dismantling of the dam, resulting in an abrupt transition from impounded to free-flowing conditions (Foley et al., 2017). While this approach initiates physical recovery almost immediately through rapid erosion and transport of accumulated sediment, substrate is often unstable and provides poor habitat in the interim (Katopodis and Aadland, 2006). In contrast, phased removal involves a gradual deconstruction process, allowing for a more controlled release of impounded sediment and potentially minimizing downstream impacts (Foley et al., 2017). Additionally, removals can be paired with restoration strategies such as artificial riffle and pool construction, channel reconfiguration, re-meandering, and bank stabilization further advancing short-term recovery (Al-Zankana et al., 2019; Katopodis and Aadland, 2006).

In a study by Claeson and Coffin (2016), a 7.9m tall and 56m wide dam was removed in a phased approach where the reservoir was drained and excavated prior to removal. The authors observed reduced macroinvertebrate abundance and richness downstream, with recovery within the first two years (Claeson and Coffin, 2016. The Payne Branch Dam removal also followed a phased approach, with additional restoration efforts including the extraction of excess sediment, channel stabilization, riffle and pool construction, and bank stabilization. Impacts on macroinvertebrates were minor, with reduced downstream total abundance, EPT abundance, EPT richness. In addition, Claeson and Coffin (2016) observed an increase in bed sediment size post-removal, that was attributed to the deposition of fines prior to removal and subsequent remobilization post-removal. Further post-removal deposition of fines was limited by excavation of the impoundment. Conversely, excavation on the Middle Fork occurred during removal, allowing for some downstream transport, likely causing the mild fining in grain size observed 3-months post removal.

Knowing that macroinvertebrates are sensitive to changes in sediment regime, particularly EPT taxa, I expected the observed differences in abundance and richness to be associated with corresponding changes in sediment dynamics, however linear regression analysis indicated there to be no significant relationship between grain size and abundance or richness. Because bed sediment size is governed by multiple factors, including source, availability, and river hydraulics, as well as the interdependent relationships among these factors, variability is easily introduced (Hack, 1957). Seasonal and climatic trends further obscure patterns, making them difficult to identify (Poff et al., 1997). That is likely the case here, as pebble counts were characterized by high variability across sites and time except at

3-months post-removal for D16 and D50 suggesting the differences observed in grain size were more consistent than for other time points.

Dam removal-induced effects on the Middle Fork were also likely dampened by the dam having been partially breached previously. Studies on dam breaches have found negative downstream impacts on aquatic biota (Gangloff et al., 2011; Maloney et al., 2008). However, rapid generation times, downstream drift, and aerial dispersal capabilities as adults are factors that have been attributed to their rapid success following disturbances (Hart et al., 2002; Sullivan et al., 2018). Researchers have reported macroinvertebrate recovery times ranging from weeks and months (Bushaw-Newton et al., 2002) to one year (Orr et al., 2008) or many years (Thompson et al., 2005). In this study, the only metrics that retained differences between upstream and downstream sites longer than six months were total and EPT abundance but may be attributed to the discharge event that occurred during the 12-month collection, rather than dam-induced effects. The timeline of recovery for the Payne Branch Dam removal is comparable to Orr et al. (2008) who studied a 2nd-order, 2.5m high, cold-water weir dam that was also partially breached. However, it is different from that of Pollard and Reed (2003) who found that the 4m high, partially breached weir dam on a 4th-order stream at the focus of their study had yet to recover after one-year post-removal or Kanehl et al. (2011) whose 4m high dam on a 4th-order river that had been previously drawn down took years to recover.

The lack of difference noted between upstream and downstream sites for diversity and biotic index might be attributed to the limited detection power of family-level identification, which may not be sensitive enough to detect subtle variations in community assemblage. Although numerous authors have evaluated the sufficiency of order and family-level taxonomy, a clear consensus has not been reached on when finer resolution is necessary. While Jones (2008) advocates strongly for taxonomic identification to the lowest level possible, Bowman and Bailey (1997) have reported little effect. More commonly, researchers suggest that the selection of taxonomic resolution should be unique to the aims of the study. Identification to family requires less time and expertise, making the work more appropriate with respect to time or financial constraints (Bailey et al., 2001). When the magnitudes of differences are expected to be large, such as differentiating an impaired and reference site, family-level may be appropriate; however, small changes like those between sites or dates in the same system can often be improved by genus and species level identification (Bailey et al., 2001; Lenat and Resh, 2001). That places dam removal studies in a gray area where it is difficult to predict the taxonomic resolution necessary to identify changes and is one limitation of this study.

5 CONCLUSION

Despite variations in recovery timelines, existing literature on dam removals indicates that the associated ecological disturbances are temporary, and vegetation, invertebrates, and fish populations recover within months to years. The results presented in this paper indicate that the Payne Branch Dam removal did not radically alter invertebrate communities, and populations made a full recovery within the first six months post-removal. Due to the unique physical, geomorphic, and ecological characteristics surrounding rivers and dams, there is no one-size-fits-all solution to river restoration. Conducting thorough pre-removal assessments to grasp the underlying ecology and seasonal variation is also crucial. In some instances, dams provide unique habitat that would be otherwise untenable, or removal of a dam can initiate further colonization by invasive species (Gangloff, 2013). The Twenty-First Century Dams Act was proposed in 2021 to address issues surrounding dam infrastructure and safety and has received bipartisan support. Additionally, the bill proposes tax credits and allocated expenditures for the removal of 1,000 dams over five years, further demonstrating the expanding need for comprehensive research (Actions - H.R.4375, 2021).

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Table 1. Locations, relative distances, and elevation of study sites. The solid horizontal line represents the dam and separates the upstream sites from the downstream sites. * Site MF1 was later excluded as an upstream reference site for macroinvertebrate metrics.

Site	Distance from Impoundment (River km)	Latitude	Longitude	Elevation (m a.s.l.)	Group
MF1	7.9	36.14489	-81.66451	1048	Upstream Reference*
MF2	3.8	36.16201	-81.64315	1015	Upstream Reference
MF3	1.6	36.17729	-81.64706	1003	Upstream Reference
MF4	0.1	36.18403	-81.65235	994	Upstream Reference
MF5	0.1	36.18531	-81.65347	989	Downstream Impaired
MF6	1.4	36.19188	-81.65491	956	Downstream Impaired
MF7	2.9	36.20209	-81.64996	943	Downstream Impaired
MF8	3.9	36.20862	-81.65376	940	Downstream Impaired

Table 2. Water chemistry parameters recorded September 12 2020 during active dam removal. The solid horizontal line represents the dam and separates the upstream sites from the downstream sites.

Site	Time	Temperature (F)	Dissolved Oxygen (%)	Specific Conductivity	Calcium	Total Dissolved Solids	pH
MF1	12:45	67	83.7	98.2	87.8	63.7	6.63
MF2	12:25	65	100	68	54	44	6.64
MF3	12:05	65.1	95.4	63.7	55.7	41.6	6.71
MF4	11:50	64.7	92.9	68.9	60.8	44.85	6.65
MF5	11:40	64.7	90.1	69.9	60.8	45.5	6.52
MF6	11:20	64.8	88.5	69.7	60.7	45.5	6.13
MF7	1:00	65.1	93.4	72.3	63.2	46.8	6.78
MF8	1:15	65.9	94.1	72.9	64.2	47.45	6.7

Table 3. North Carolina Biotic Index bioclassifications using Small Stream Criteria for the Mountain ecoregion. The solid vertical line represents the dam and separates the upstream sites from the downstream sites.

Months Post-Removal	MF1	MF2	MF3	MF4	MF5	MF6	MF7	MF8
0	Good	Good	Good	Good	Good	Good	Good	Excellent
1	Good	Good	Good	Good	Good	Good	Good	Good
2	Good	Good	Good	Good	Good	Good	Good	Good
3	Good	Good	Good	Good	Good	Good	Good	Good
4	Good	Good	Good	Good	Good-Fair	Good	Good	Good
5	Good	Good	Good	Good	Good	Good	Good	Good
6	Excellent	Good	Good	Good	Good	Good	Good	Good
9	Good	Good	Good	Excellent	Good	Excellent	Good	Good
12	Good	Good	Good	Good	Excellent	Excellent	Excellent	Good

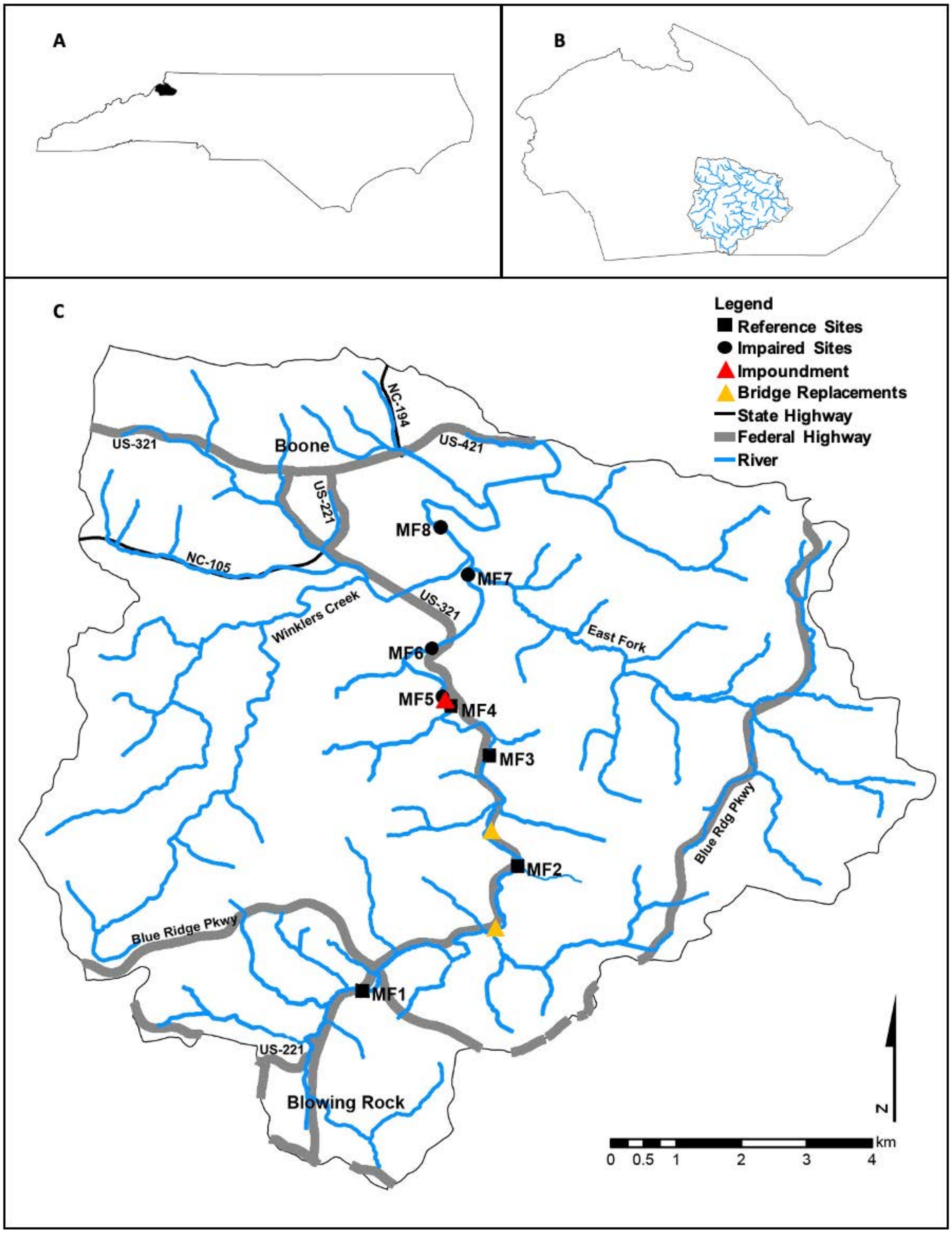

Figure 1. Maps depicting A) Watauga County, North Carolina, B) the Headwaters South Fork Watershed of the Upper New River subbasin within Watauga County, and C) the study area and relative locations of collection sites. Sites MF1, MF2, MF3, MF4, MF5, MF6, and MF7 are located on the Middle Fork of the South Fork New River, whereas MF8 is located on the South Fork New River, downstream of the confluence of the Middle Fork, East Fork, and Winklers Creek.

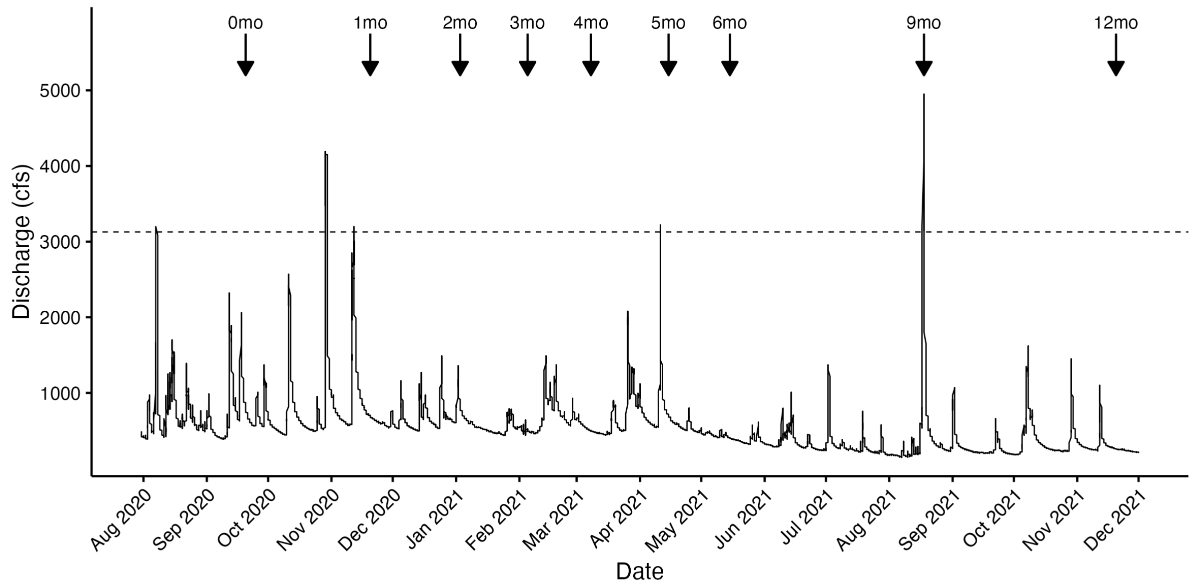

Figure 2. Discharge of the South Fork of the New River near Jefferson, NC, approximately 80 river km downstream of the study area. The dashed line indicates 3,128cfs and corresponds to discharge at the depth in which the river reaches flood stage. Arrows represent approximate macroinvertebrate and sediment collection dates. Data retrieved from the U.S. Geological Survey (USGS Water Data Services, 2023).

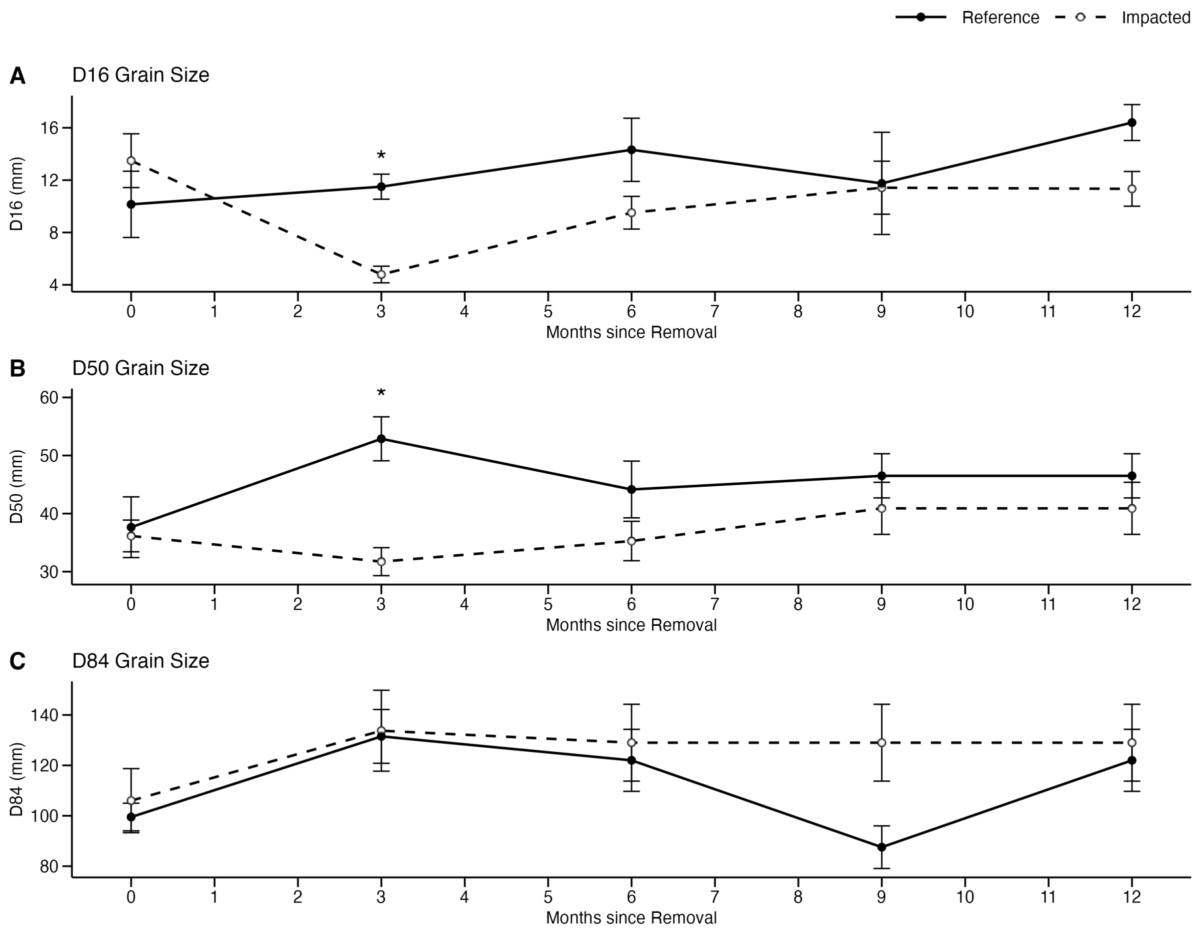

Figure 3. Temporal changes in sediment grain size distribution for the A) 16th percentile (D16), B) 50th percentile (D50), and C) 84th percentile (D84) following dam removal. The x-axis represents time in months post-removal, while the y-axis shows the mean grain size. Error bars extending above and below each data point represent the standard error of the mean (SEM). Significant differences between reference and impaired sites are indicated *.

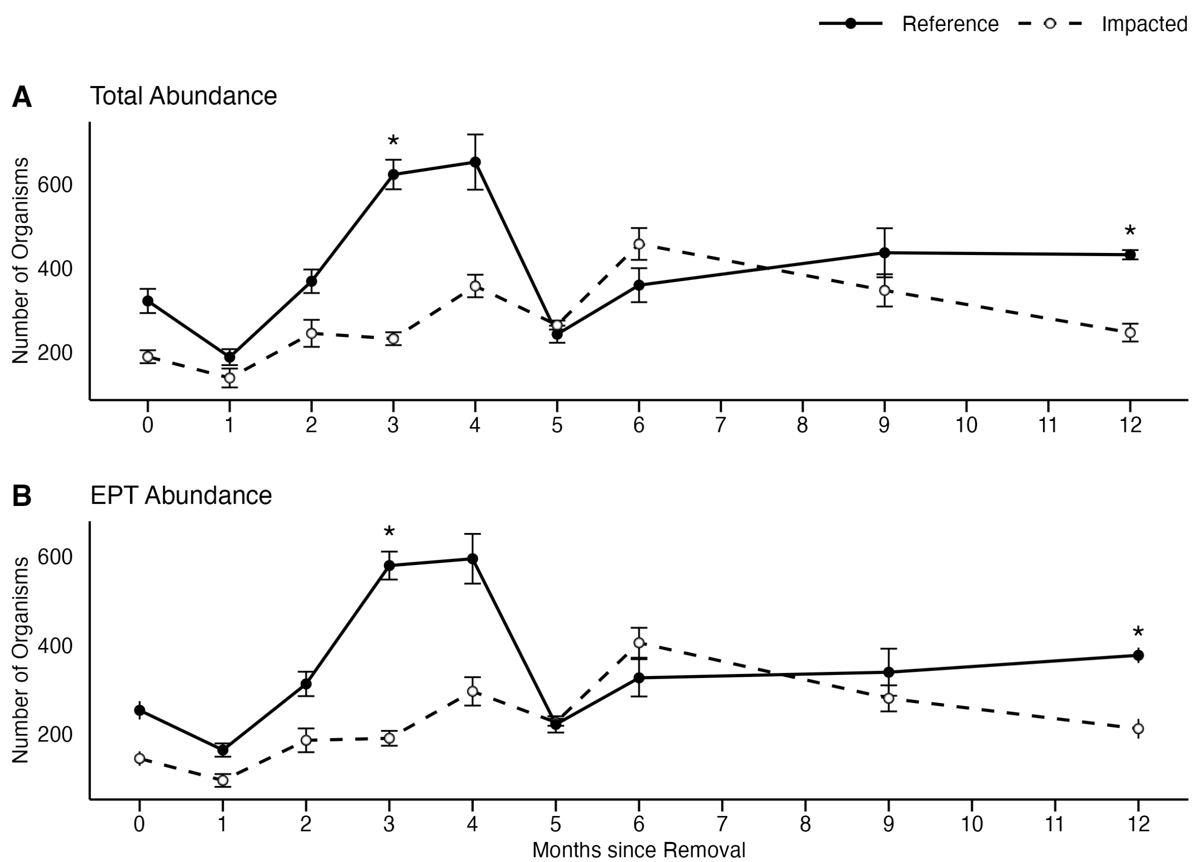

Figure 4. Temporal changes in A) total abundance and B) EPT abundance of macroinvertebrates following dam removal. The x-axis represents time in months post-removal, while the y-axis shows the mean number of organisms. Error bars extending above and below each data point represent the standard error of the mean (SEM). Significant differences between reference and impaired sites are indicated *.

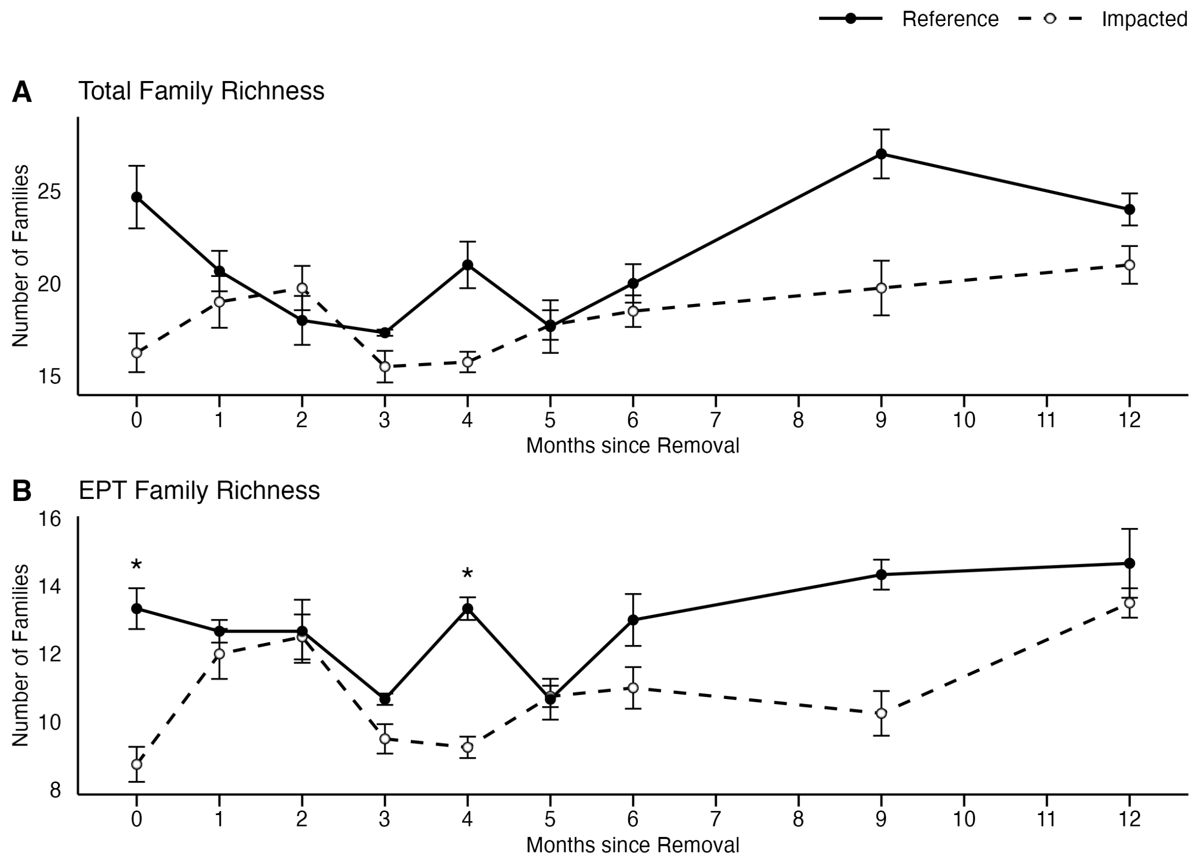

Figure 5. Temporal changes in A) total family richness and B) EPT family richness of macroinvertebrates following dam removal. The x-axis represents time in months post-removal, while the y-axis shows the mean number of organisms. Error bars extending above and below each data point represent the standard error of the mean (SEM). Significant differences between reference and impaired sites are indicated *.

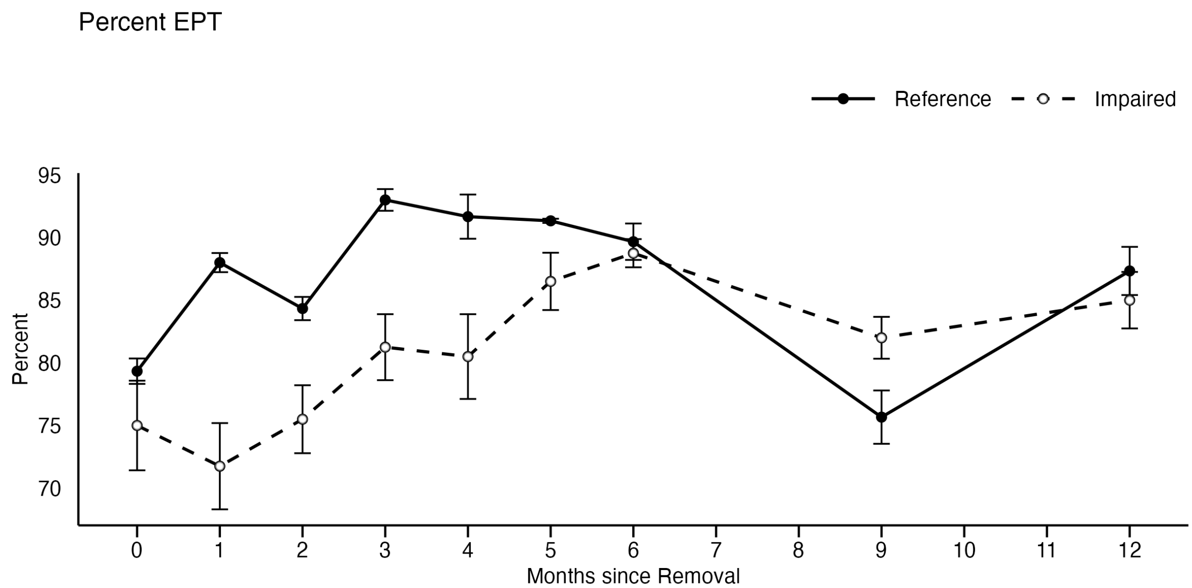

Figure 6. Temporal changes in percent EPT following dam removal. The x-axis represents time in months post-removal, while the y-axis shows the mean percent. Error bars extending above and below each data point represent the standard error of the mean (SEM). No significant differences between reference and impaired sites were detected.

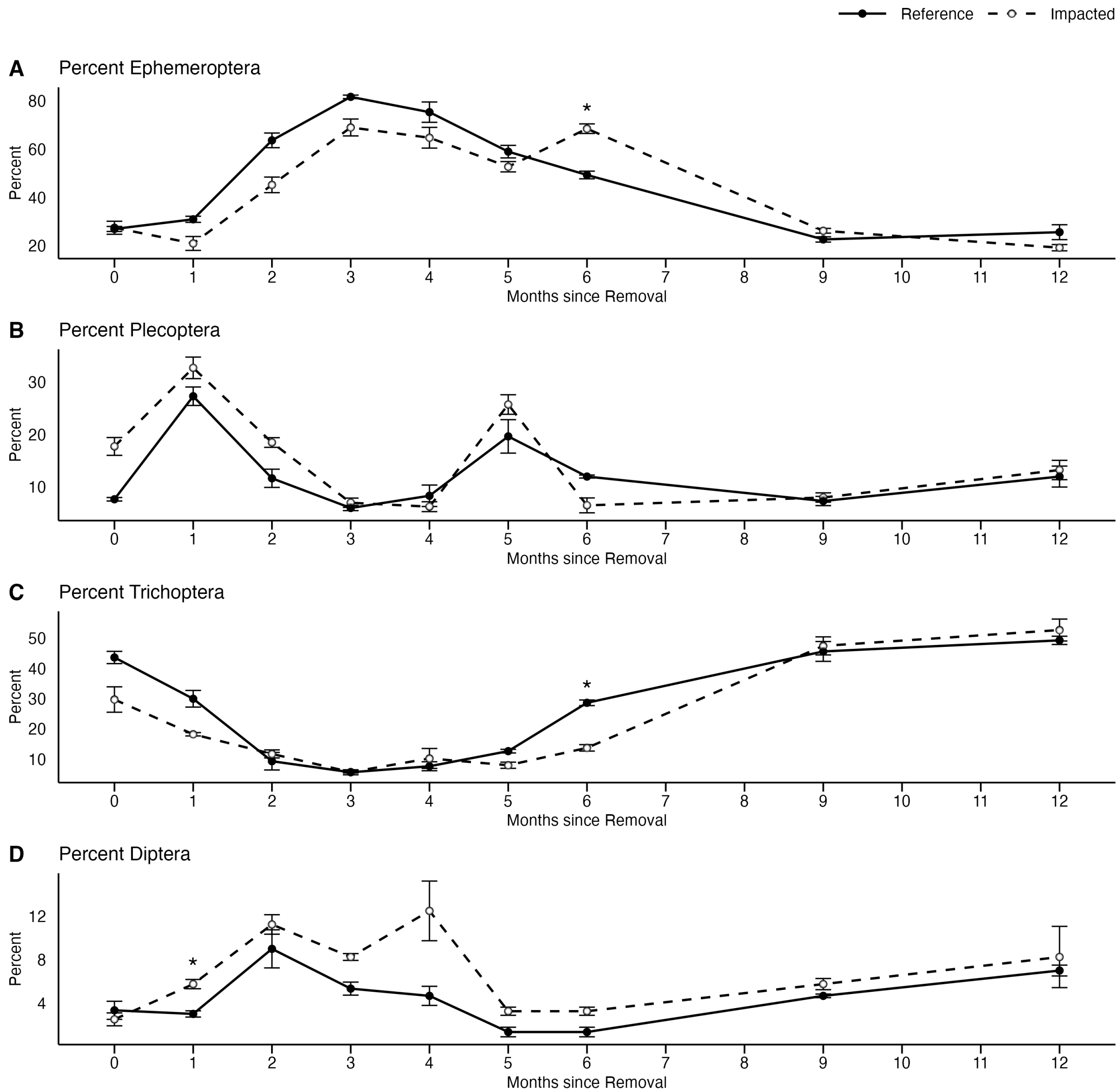

Figure 7. Temporal changes in A) percent Ephemeroptera, B) percent Plecoptera, C) percent Trichoptera, and D) percent Diptera following dam removal. The x-axis represents time in months post-removal, while the y-axis shows the mean percent. Error bars extending above and below each data point represent the standard error of the mean (SEM). Significant differences between reference and impaired sites are indicated *.

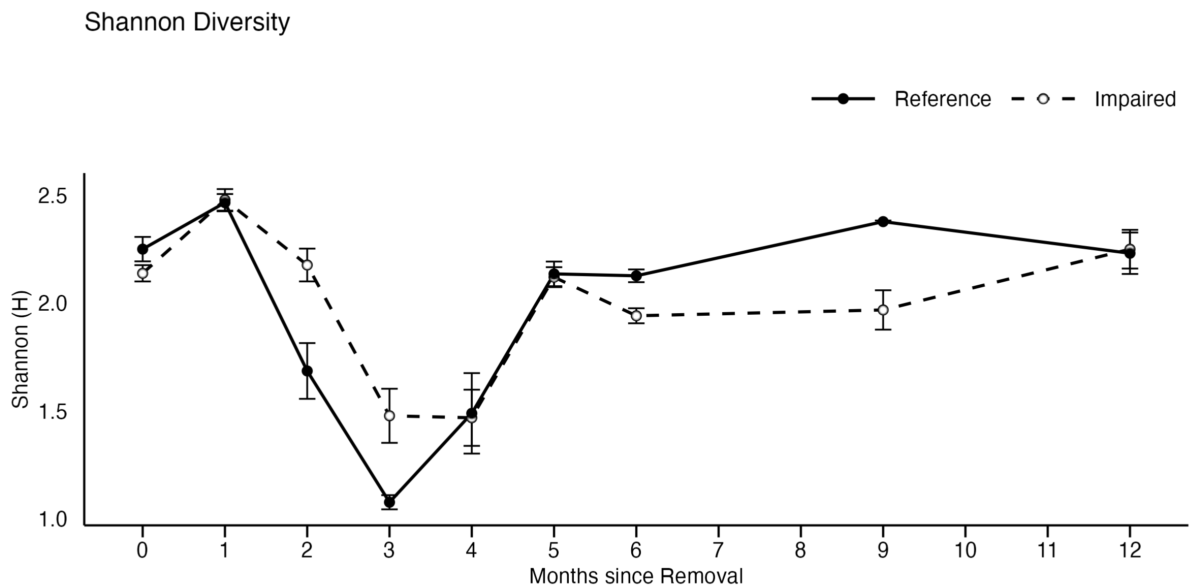

Figure 8. Temporal changes in Shannon-Weiner diversity index of macroinvertebrates following dam removal. The x-axis represents time in months post-removal, while the y-axis shows the mean Shannon diversity. Error bars extending above and below each data point represent the standard error of the mean (SEM). No significant differences between reference and impaired sites were detected.

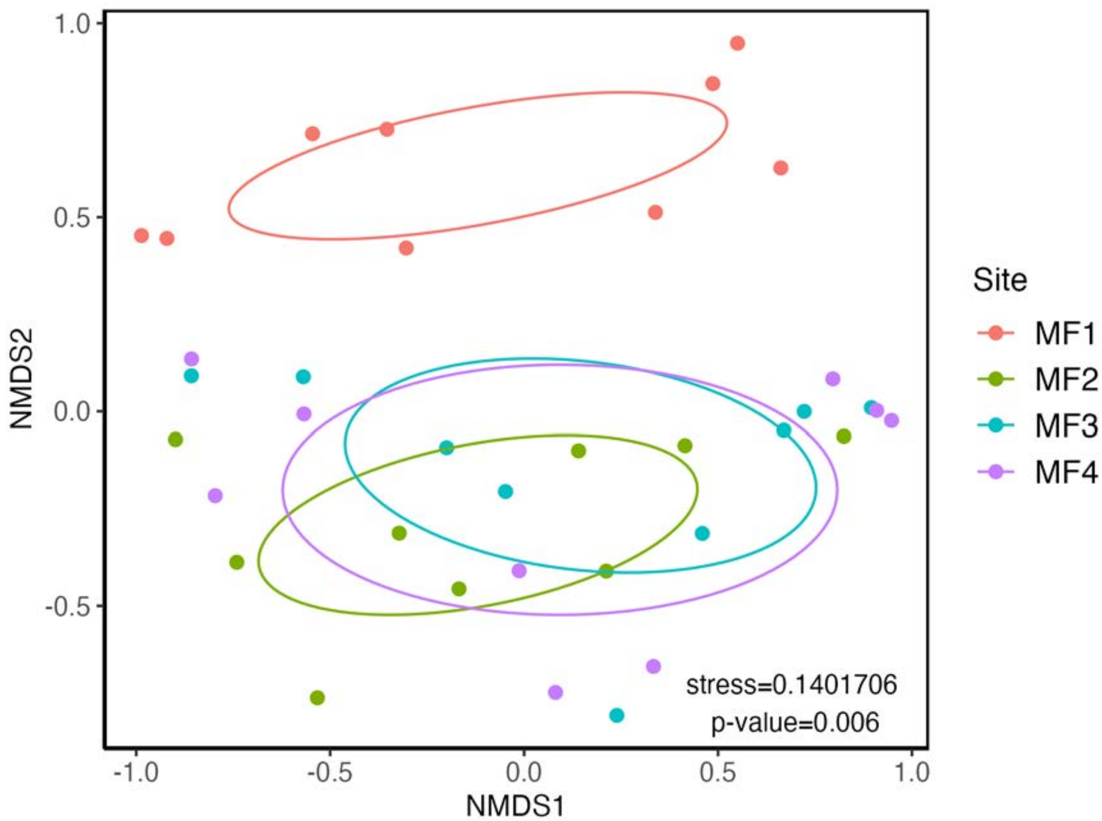

Figure 9. Non-Metric Multidimensional Scaling (NMDS) plot visualizes the Morisita-Horn dissimilarity of upstream study sites based on a family-level macroinvertebrate abundance matrix. The stress value quantifies the goodness of fit. PERMANOVA yielded a p-value of 0.006. In this plot, each point represents an individual study site, and the spatial arrangement of points reflects the degree of similarity in macroinvertebrate community composition. Ellipses are drawn around groups of points to indicate the spread or variation within each group.

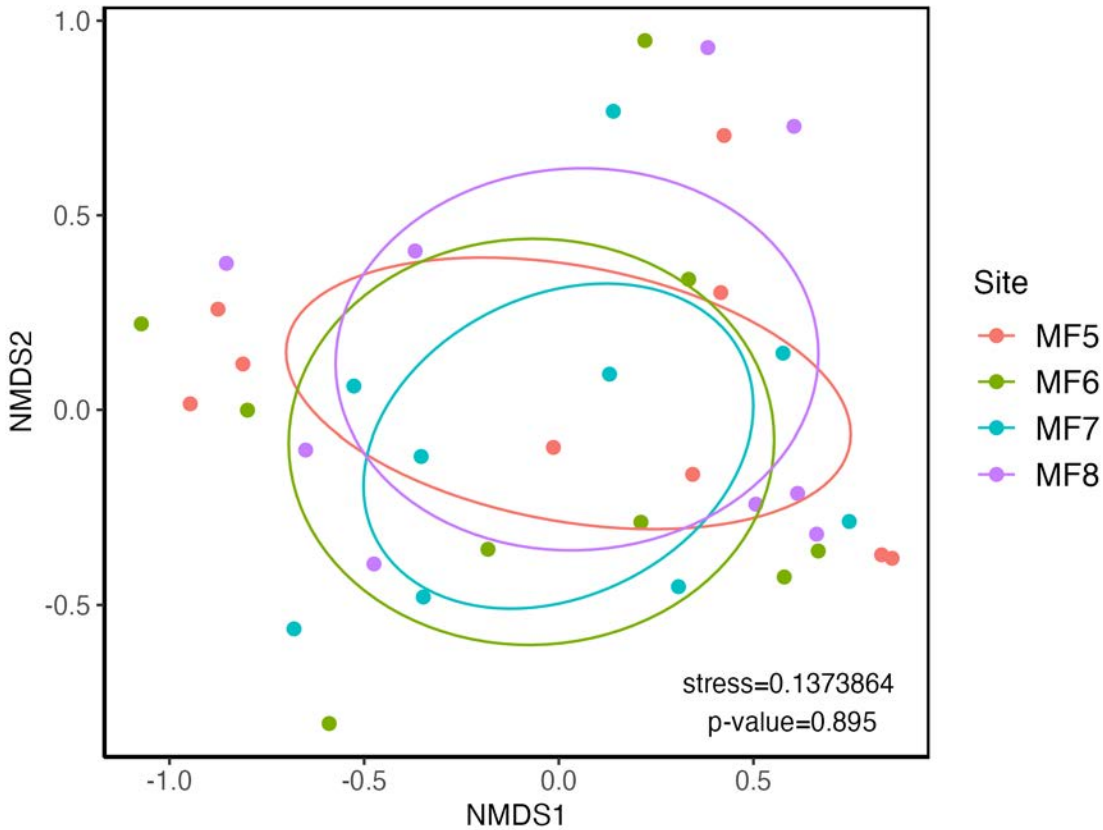

Figure 10. Non-Metric Multidimensional Scaling (NMDS) plot visualizes the Morisita-Horn dissimilarity of downstream study sites based on a family-level macroinvertebrate abundance matrix. The stress value quantifies the goodness of fit. PERMANOVA yielded a p-value of 0.895. In this plot, each point represents an individual study site, and the spatial arrangement of points reflects the degree of similarity in macroinvertebrate community composition for all sampling dates. Ellipses are drawn around groups of points to indicate the spread or variation within each group.

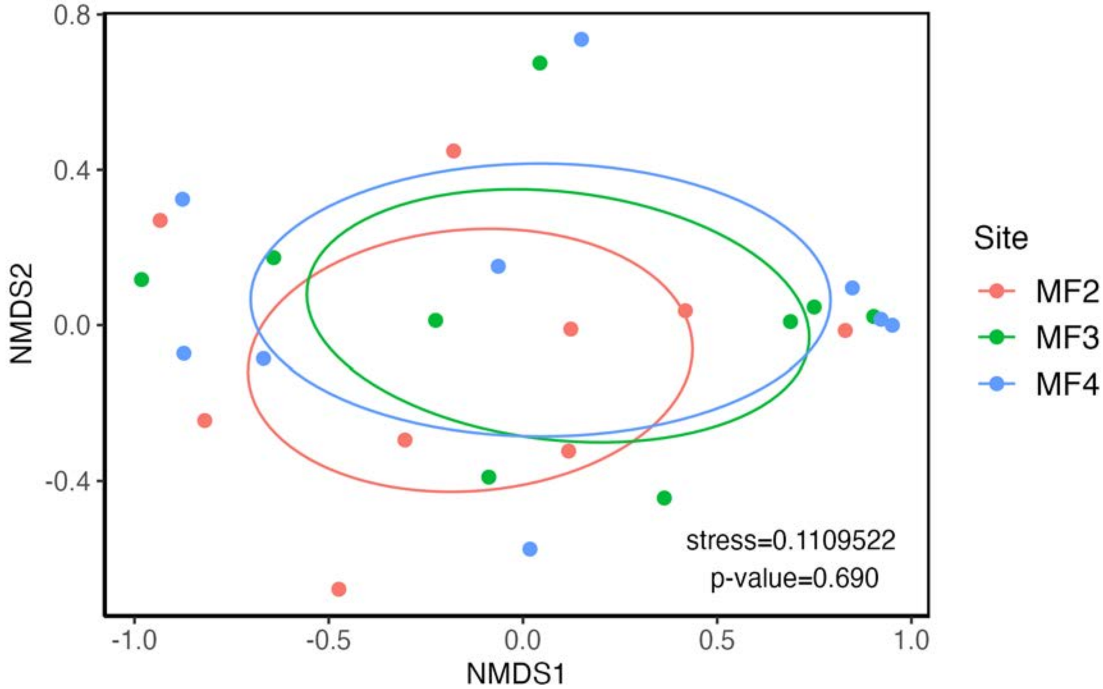

Figure 11. Non-Metric Multidimensional Scaling (NMDS) plot visualizes the Morisita-Horn dissimilarity of upstream study sites, excluding MF1, based on a family-level macroinvertebrate abundance matrix. The stress value quantifies the goodness of fit. PERMANOVA yielded a p-value of 0.690. In this plot, each point represents an individual study site, and the spatial arrangement of points reflects the degree of similarity in macroinvertebrate community composition. Ellipses are drawn around groups of points to indicate the spread or variation within each group.

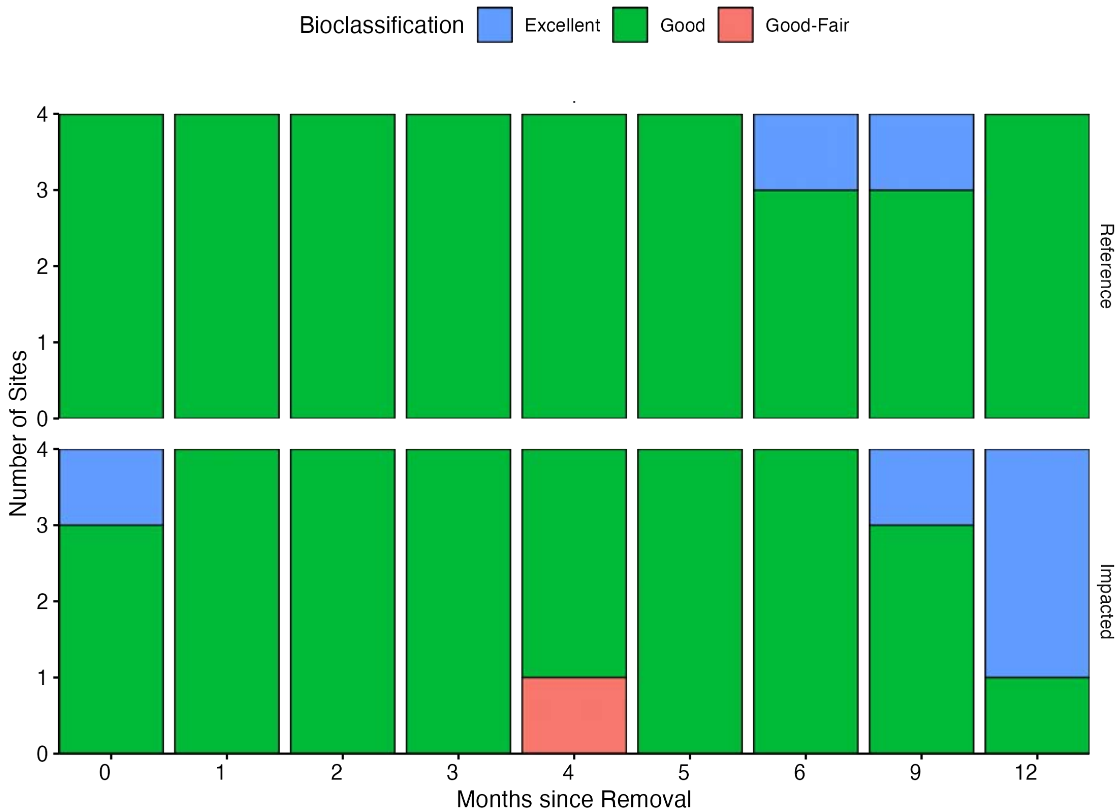

Figure 12. Temporal changes in North Carolina Biotic Index bioclassifications following dam removal. NCBI was calculated using Small Stream Criteria for the Mountain ecoregion. The x-axis represents time in months post-removal, while the y-axis shows the number of sites with a given bioclassification for reference and impacted communities.

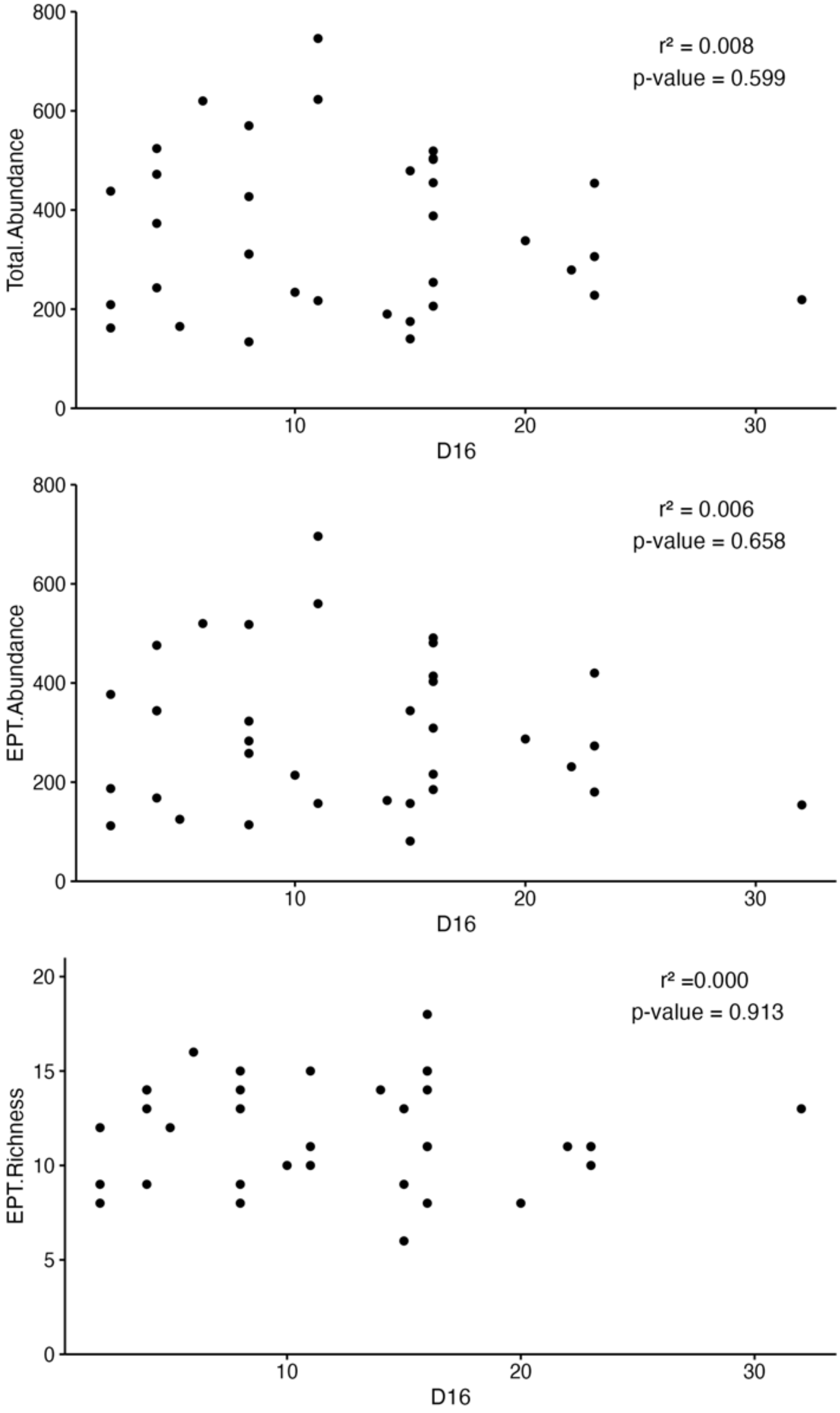

Figure 13. Linear regression analysis using one-way ANOVA model depicting the relationship between D16 grain size and total abundance, EPT abundance, and EPT richness. No significant correlations were observed.

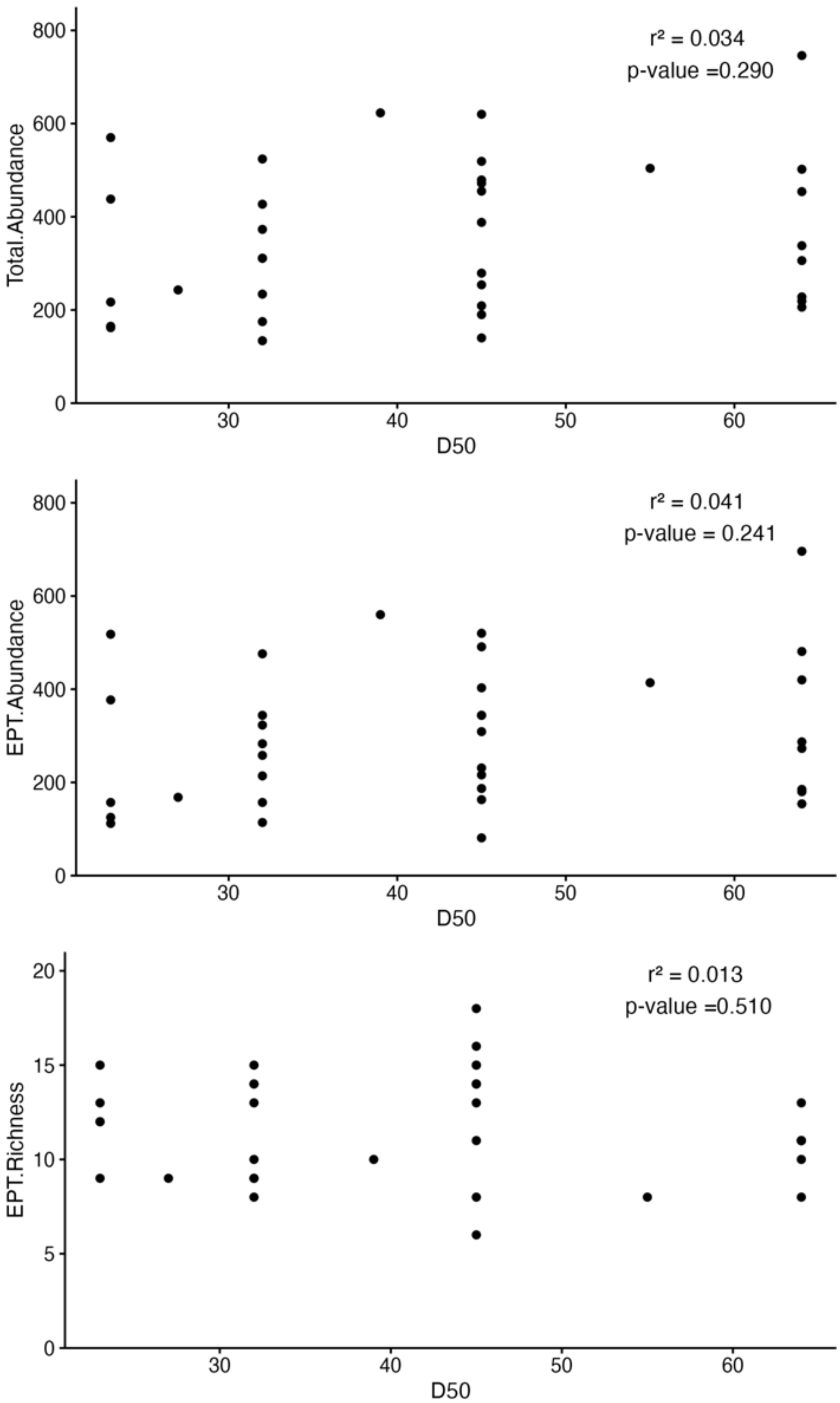

Figure 14. Linear regression analysis using one-way ANOVA model depicting the relationship between D50 grain size and total abundance, EPT abundance, and EPT richness. No significant correlations were observed.

Vita

Madison Blue Suttman was born in Rockford, Illinois to Benjamin Suttman and Adriann Mangrum. She graduated from Dakota Jr/Sr High School in May of 2016. In May of 2020, she graduated from Beloit College in Beloit, Wisconsin with a Bachelor of Science in Biology and Environmental Justice and Communication. After graduation, she moved to Boone, North Carolina to attend Appalachian State University under the guidance of Dr. Shea Tuberty and was awarded a Master of Science in Biology in December of 2023. She is now attending the University of Illinois Urbana-Champaign for a Ph.D. in Natural Resources and Environmental Sciences. In her free time Madison enjoys photography, crafting, hiking, and skiing.