MULTIDISCIPLINARY ASSESSMENT OF THE REMOVAL OF A SMALL RUN-OF-RIVER DAM IN THE SOUTHERN APPALACHIAN MOUNTAINS

A Thesis
by
WILLIAM GREGG MCMAHAN

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APPROVED BY:

Dr. Shea Tuberty
Chairperson, Thesis Committee

Dr. William Armstrong
Member, Thesis Committee

Dr. Michael Gangloff
Member, Thesis Committee

Dr. Derek Martin
Member, Thesis Committee

Dr. Ava Udyadia
Chairperson, Department of Biology

Dr. Marie Hoepfl
Interim Dean, Cratis D. Williams School of Graduate Studies

Abstract

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William Gregg McMahan
B.S. Appalachian State University
M.S. Appalachian State University

Chairperson: Dr. Shea Tuberty

Dam removals are a commonly employed tool of water resource managers for stream restoration projects. Most dam removals are performed without the accompaniment of a scientific study to evaluate their efficacy or biological consequences, and many of the ones which are studied are situated in low gradient, warmwater streams. The removal of the Ward Mill Dam on the Watauga River in the Appalachian Mountains of North Carolina provided an ideal case study for a dam removal in a moderate gradient, coolwater stream. Benthic macroinvertebrate samples were collected and sediments surveyed from 8 sites along the Watauga River for 6 months following the dam removal to identify the downstream extent and temporal persistence of changes to the benthic community. Sites further than 1 km downstream of the dam showed moderate to extreme long-lasting alterations to the median streambed particle size, but few changes to the benthic community. Sites within the former impoundment and immediately downstream of the dam experienced drastic changes to channel morphology and macroinvertebrate community structure. At 6 months post removal nearly all benthic community metrics had recovered to reference condition. This suggests that headwater streams may recover more slowly to geomorphic changes following dam removals, however benthic macroinvertebrate communities rebound quickly to refill the vacated ecological roles following the disturbance.

Acknowledgements

I would like to thank Madison Suttman and Jacob Cunningham for their help in the field, I would not have been able to complete this project without your dedicated assistance. I would also like to thank Josh Platt for his detailed work on the geomorphology of the Watauga River in regard to the Ward Mill Dam removal, as well as for sharing his photographs of the dam removal. My father, Gregory McMahan, helped considerably with the data digitization and processing. His contributions were essential to the analysis of abiotic responses to the dam removal. Finally and most importantly, I would like to thank the Ward Family. Their interest in restoring this beautiful river to its natural flow and continued cooperation with Appalachian State University researchers enabled the dam removal to occur and made empirical study of the results possible.

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Foreword

This thesis manuscript will be submitted to the *Journal of the American Water Resources Association (JAWRA)*. The thesis has been formatted according to the style guide for this journal for minimal revisions before publication.

Background

The Ward Mill Dam was a small (6 m tall) timber and concrete dam on a 5^{th} order reach of the Watauga River near the community of Sugar Grove, North Carolina, USA initially constructed of hemlock logs in 1905 (Wigginton, 1980). The dam was the only one in Watauga County, North Carolina to survive the 1940 flood, when 20 cm of rain fell in 48 hours onto already saturated soil, washing out roads, dams, houses, and railroad tracks (NRLP, 2021; Osment, 2008; Wigginton, 1980). A hydropower turbine was installed in the 1930s, and in 1963 the structure was reinforced with concrete and a fish ladder was installed (Wigginton, 1980). The impounded reach extended 700 m upstream through a narrowing of the valley and stored 20,000 m^3 of water at full pool (FERC, 2014). The Ward family filed to renew the operating license in 2014 and the dam was approved for continued operation in 2017; however requirements involved in the continued operation of the dam prompted the Ward family to surrender the license and remove the dam (FERC, 2017). Dam removal was conducted by the Unites States Fish and Wildlife Service (USFWS) in a single operation which began on 16 May 2021 and took two days to complete.

Introduction

The United States Army Corps of Engineers (USACE) monitors more than 90,000 dams greater than 7.6 m (25 ft) tall across the US, with most of them built in the Southeast and Plains States (USACE, 2022). More than 1600 dams have been removed across the US, and the rate of removal is increasing (American Rivers, 2019). The majority of dam removals in the US are small dams $<$ 10 m tall and most removals have occurred in the Northeast, Upper Midwest, or Pacific Coast (Foley *et al.*, 2017a). Fewer than 10% of these removal projects are empirically studied, and most of these studies are of limited duration following the removal and occur in warm water, low gradient streams (Bellmore *et al.*, 2017; Gillette *et al.*, 2016). The location of Ward Mill Dam on a moderate gradient (16° basin average) coolwater river will provide important information to this poorly studied removal environment (Lyons *et al.*, 2009).

Man-made dams alter the natural continuity of river systems; acting as traps for sediment and woody debris, fragmenting lotic ecosystems, altering downstream geomorphology and habitat, and altering flow regimes (Bednarek, 2001; Poff and Hart, 2002; Tullos *et al.*, 2014). The bedload and suspended sediment transported by rivers settles within the impoundment of large dams, which leaves the downstream reaches starved of sediment and can lead to ero- sion (Collier *et al.*, 1996; Graf, 2006). The hydrology of run-of-river dams, such as at Ward Mill, is more complicated with shorter hydraulic residence times and variable velocity through the backwater (Csiki and Rhoads, 2010). During low flow periods, bedload and suspended sediments settle out within the impoundment whereas at high flow stages the increased shear stresses entrain fine particles and carry them over the dam crest as washload (Csiki and Rhoads, 2010). The ability of streams to transport bedload material depends on the specific properties of the dam and velocity of the turbulence currents. Dams fragment riverine ecosystems through physical obstruction of migration routes and physiochemical changes to habitats. Disruption of longitudinal distribution of riverine fish can isolate populations, increasing the risk of local population decline due to limiting access to external gene flow, spawning grounds, or feeding areas (Helms *et al.*, 2011; Ward and Stanford, 1983). Dams also impede or prevent the migration of anadromous fish, even with the installation of costly fish bypass systems (Winter, 1990). Artificial impoundments can also serve as toe-holds for the establishment of non-native aquatic organisms (Johnson *et al.*, 2008).

Reasons for Removals

Dams are ubiquitous across the United States, with 91,752 recognized by USACE (USACE, 2022). Many of the larger hydroelectric, water supply, or flood control dams were built by the U.S. Bureau of Reclamation and USACE during the 1960s, but the smaller dams were primarily built in the 1800s and early 1900s to provide water for mills, industry, or water supply (Csiki and Rhoads, 2010; Reisner, 1986). These dams are meeting or have exceeded their engineered lifetimes, with rising maintenance costs and few explicit decommissioning plans (Doyle *et al.*, 2008). A number of these smaller dams no longer fulfil their original purpose and have fallen into disrepair, increasing the potential for failure (Pejchar and Warner, 2001). Small run-of-river dams additionally pose a drowning hazard to recreational river users, creating a dangerous "drowning machine" recirculating hydraulic jump downstream of the dam which can trap swimmers and boats (Treinish, 2017). In addition to economic or public safety risks and hazards, dams are increasingly being removed for ecological reasons (Foley *et al.*, 2017a). River managers remove dams to restore flow and sediment regimes to a more natural condition and to facilitate the recovery of native organisms into formerly fragmented habitats (Bellmore *et al.*, 2017).

Impacts of Removals

Dam removals have been occurring at an increasing rate across the United States, however empirical research on the responses of stream habitats and biota has been unable to keep pace (American Rivers, 2019; Ding *et al.*, 2019). There is high variability across the dam removal study literature of hydrologic setting, sediment load, local biological community, dam construction, and removal tactics. The emerging trends of dam removal research indicate that there are numerous changes to the stream channel, and both positive and negative impacts to the ecological communities downstream, which stabilize to a new steady-state equilibrium along varying trajectories and timelines (Bellmore *et al.*, 2017; Foley *et al.*, 2017a).

Breaching of the dam immediately releases a pulse of sediments downstream, which can scour or bury downstream habitats depending on local conditions, even locally extirpating freshwater mussel colonies through the shift from gravel substrates to sandy bedforms (Gangloff, 2013; Hart *et al.*, 2002). Downstream communities of periphyton, fish, and benthic macroinvertebrates are negatively affected by the sediment slug, resulting in decreased total macroinvertebrate abundance and relative abundance of sensitive taxa such as Ephemeroptera, Plecoptera, and Trichoptera (collectively EPT) (Carlson *et al.*, 2018). Experimental applications of fine sediment into benthic trays revealed that sedimentation reduces taxa richness and density, leads to a decrease in trait diversity, and a shift to more sediment resistant taxa (Angradi, 1999; Larsen *et al.*, 2011). However, drawdown of the lake level following dam removal can expose submerged and buried riffles within the former impoundment, which are colonized rapidly by benthic invertebrates that drift from upstream sites, particularly hydropsychid caddisflies (Cook and Sullivan, 2018). Downstream benthic invertebrate communities shift to a *r*-selected, disturbance and pollution tolerant community structure following dam removal, but recover to resemble the upstream communities within months following the removal (Chiu *et al.*, 2013; Tullos *et al.*, 2014).

The shift from lotic habitat to lentic habitat in the impoundment also leads to a change in fish assemblage upstream of the dam, and the physical obstruction fragments migratory populations (Burroughs *et al.*, 2010). Decreased current velocity and sediment transport ability upstream of the dam creates backwater habitat with lower dissolved oxygen, increased likelihood of thermal stratification, and a reduction in substrate size and suitable spawning condition (Ward and Stanford, 1983). Downstream of the dam, increased current velocity and sediment transport capacity causes scour of smaller particles and an increase in substrate size which leads to loss of suitable spawning conditions (Csiki and Rhoads, 2010; Kondolf, 1997). Removal of a dam mobilizes the trapped sediments, redistributing them downstream of the former impoundment and on the adjacent floodplain, where an increased load of fine sediment buries deepwater habitats (Burroughs *et al.*, 2010; Tullos *et al.*, 2014). Increased turbidity during the sediment pulse transport lowers visibility in the water, negatively impacting visual predators such as salmonids (Larsen *et al.*, 2011). The transient nature of increased turbidity is likely insufficient to reduce photosynthesis and primary productivity, however decreases in downstream algal communities have been detected and attributed to burial or scour (Orr *et al.*, 2008). Within watersheds with non-native invasive taxa, the removal of a dam can have deleterious effects by allowing the invaders to spread past their previous confinement (Jackson and Pringle, 2010). Fish recovery time and recovery trajectory following a dam removal is dependent on the specific needs of the fish taxa in question, amount and erodibility of the accumulated sediments, and amount of geomorphic adjustment required to satisfy habitat preferences of fish taxa (Doyle *et al.*, 2005).

The study of a breached dam that was later removed showed no change to the fish communities before and after the removal, likely because no difference was found upstream and downstream of the breached dam before removal indicating that the communities had already recovered following the breach in 1960 (Gillette *et al.*, 2016). Following the removal of Edwards Dam on the Kennebec River, Maine, thousands of Alewives (*Alosa pseudoharengus*), Striped Bass (*Morone saxatilis*), and Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) passed the site of the former dam for the first time in over 100 years, with the hope that the Atlantic Salmon (*Salmo salar*) will follow (Crane, 2009). Surveys of local fishermen indicated that they value the restored fishery by their willingness to spend more money to visit the fishery and to improve access and facilities at the fishery (Robbins and Lewis, 2008). Colleagues in the Appalachian State University department of Biology studying the fish response to the Ward Mill Dam removal found that the dam removal predominately influenced fish communities within the former impoundment and tailrace, along with the free flowing reach immediatley upstream of the impoundment; where decreases in richness were found downstream of the dam and increases in richness were found in the impoundment and immediately upstream of the impoundment ((Gangloff *et al.*, 2022)).

Seasonality and Climatic Impacts on Dam Removal Projects

The life histories of many aquatic organisms are adapted to seasonal disturbances such as floods (Tullos *et al.*, 2014). Predictable seasonal floods present as a regularly occurring pulse of turbidity, benthic scour, and sediment deposition, and aquatic organisms will have adapted traits to alleviate the effects of the disturbance, such as entering an aerial adult phase or diapause to avoid the flood (Lytle and Poff, 2004). When managers time dam removals to the start of flood season in a predictable hydrologic setting, the hypothesis is that the downstream benthic community will be adapted to seasonal increases in sediment and may be less impacted by the removal (Chang *et al.*, 2017).

Models of sediment response to dam removals developed following more than a decade of dam removal research suggest that the impoundment is excavated of stored sediment through a two phase process where the initial excavation is driven by local base level fall with subsequent excavation by flood events (Collins *et al.*, 2017; Pearson *et al.*, 2011). The removal of the Merrimack Village Dam in 2008 and the removal of the Simkins Dam in 2010 allowed for paired studies investigating channel response following sediment pulses in two hydrologically similar settings (Collins *et al.*, 2017; Pearson *et al.*, 2011). The rapid phase of recovery within the impoundment follows the proposed initial stages of the earlier channel evolution model of rapid incision, followed by slower incision and associated widening (Collins *et al.*, 2017; Pearson *et al.*, 2011). The aggradation of the channel from upstream sediment and stability of the floodplain proposed by the early model are achieved during the subsequent event-driven stage of recovery (East *et al.*, 2018). Early models predicted a continuous recovery, however measurements of erosion fit poorly with a single exponential decay (Pearson *et al.*, 2011). A two stage decay, with a single break point separating two distinct decay constants fit the data more closely (Figure 1).

A single large seasonal flood post removal can potentially excavate the legacy sediments from former impoundments, increasing the sediment throughput rate of the downstream reaches (Bednarek, 2001; Kondolf, 1997). Since record keeping began floods have increased in frequency and magnitude across New England and the eastern United States, with a stepped increase found around the year 1970 in many streams (Armstrong *et al.*, 2014; Groisman *et al.*, 2001). Benthic macroinvertebrates evolved in environments with frequent small floods, and the substrate characteristics established by that flow regime (Chiu and Kuo, 2012). Stream channels are maintained by low intensity floods occurring once or twice per year, as these carry more than 50% of the annual suspended sediment load (Wolman and Miller, 1960). Increased magnitude and frequency of flood flows suggest that eastern rivers may shift to accommodate the new hydrologic conditions, altering benthic habitats in the process.

Implications of Study

Studies of small dam removals on moderate gradient coolwater montane streams in the Southeastern United States are rare in the scientific literature (Gillette *et al.*, 2016; Foley *et al.*, 2017a). The scheduled removal of the Ward Mill Dam on the Watauga River in Western North Carolina was fortunate due to the proximity to Appalachian State University and that advanced planning for the dam removal allowed for pre-removal sampling. This study, which is part of a larger collaborative effort with the Department of Geography and Planning, the Department of Geological and Environmental Sciences, and other researchers within the Department of Biology, was intended to provide new information in an understudied dam removal scenario.

Within this aspect of the study the objectives were to quantify the extent of dam removal dependent depletion or adaptation of the benthic invertebrate communities, determine the spatial extent of BMI depletion, measure the rate of recovery for the BMI community, describe the seasonal flow variability and flood trends of the Watauga River at this site, and to describe methods for predicting flood seasonality on future dam removal projects. It was expected that the downstream sites would experience aggradation and an increase in percent fines. This was in turn expected to precipitate a short-term shift in benthic macroinvertebrate communities to an increased percent of *r*-selected taxa, with reductions in scrapers and clingers. Recovery at Impacted sites to Upstream Reference condition was expected to occur within 6 months post removal. This information will help regulatory agencies and interest groups plan future removals within the Southeast, where numerous aging dams continue to impound the regions rivers (Bellmore *et al.*, 2017).

Field Site Description

A modified Before-After-Control-Impact (BACI) study design was utilized to determine response to the dam removal and recovery timeline. Camp Broadstone, NC Highway 194 bridge, Rominger Road bridge, and US Highway 321 bridge were intended to serve as reference sites, while Ward Mill Impoundment, Ward Mill Tailrace, Pasture Site, and Hubert Thomas Road bridge were intended to serve as impacted sites (Table 1, Figure 2). Macroinvertebrate responses to the dam removal at the Pasture site and Hubert Thomas Rd. bridge were more similar to the US Hwy. 321 bridge than to the Ward Mill Impoundment and Ward Mill Tailrace during preliminary analysis. In order to capture this diversity among impacted sites the study design was modified to include one reference group and two impacted groups for finer resolution. Camp Broadstone, NC Hwy. 194 Bridge, and Rominger Rd. bridge serve as Upstream Reference sites, Ward Mill Impoundment and Ward Mill Tailrace serve as Dam Impacted Sites, and Pasture Site, Hubert Thomas Rd. bridge, and US Hwy. 321 bridge serve as Downstream Impacted Sites.

Camp Broadstone is located at the upstream end of Valle Crucis, with bedrock and boulder confinement on the right bank and floodplains on the left bank, and has a deep channel dominated by bedrock and boulders (Figure 3). North Carolina Hwy. 194 bridge crosses the Watauga River at the confluence with Dutch Creek, just downstream of the site named for the bridge (Figure 4). The site is a gravel and cobble channel in an unconfined section of Valle Crucis with a sloped cobble bank on river left and a steep right bank of consolidated fines supported by included cobbles. Rominger Rd. crosses the Watauga River downstream of the confluence with Cove Creek near the downstream end of Valle Crucis, and the Rominger Rd. bridge site is located upstream of the confluence (Figure 5). The channel is composed of cobbles and boulders, with large (2-4 m) boulders confining the left bank and a floodplain on the right bank. The Impoundment Site was located within the former Ward Mill impoundment (Figure 6). The site changed dramatically throughout the course of the study, beginning as a lentic environment with a mud bed, changing to a sand bed following the removal, and coarsening to a gravel bed with isolated boulders (Figure 7). The Tailrace Site was located immediately downstream of the former Ward Mill Dam (Figure 8). This site changed throughout the study, beginning as a cobble and boulder bed confined by cliffs on river right and boulders on river left, being buried by sand following the dam removal and then coarsening to a gravel and cobble bed (Figure 9). The Pasture Site is located 1 river km downstream of the dam, near where the road once again meets the river after bypassing a steep drainage (Figure 10). The channel through this site has a floodplain on the left bank and is confined on the right bank by bedrock. Prior to removal the bed at the Pasture site was gravel and cobbles, which infilled with sand following the removal and was buried by sand and gravel following the flood on 16 August 2021 (Figure 11). Hubert Thomas Rd. crosses the Watauga River with a low water bridge and the Hubert Thomas Rd. bridge site is located immediately downstream of the bridge (Figure 12). The channel is un-confined and composed of gravel and cobbles with isolated boulders. Following the removal infilling with sand occurred without visible alterations to the channel. United States Hwy. 321 crosses the Watauga River at a sharp bend with a public river access (Figure 13). The site is located upstream of the bridge at the confluence with an unnamed tributary on the right bank. The channel at this site has a floodplain on river right and is confined by valley walls on river left, and transitions from gravel and cobbles in the upstream half to cobbles and boulders in the downstream half, with no notable change to the D_{50} or visual characteristic throughout the study period.

Methods

Hydraulic Characterization and Flood Trends

Peak annual discharge between 1940 and 2021 along with an estimated peak annual discharge for the 1916 flood were obtained from the United States Geological Service (USGS) to create a recurrence interval model for the Watauga River (USGS, 2022). The lowest peak annual discharge on record, 1700 cubic feet per second (cfs), occurred in 1999 and is the one-year flood magnitude. Discharge data were natural log-transformed and ranked in descending order to calculate recurrence intervals for the remaining recorded events. Recurrence intervals (RI) were calculated using Equation 1:

$$RI = (n + 1)/m \tag{1}$$

where n is the number of events on record and m is the magnitude rank (IACWD, 1981). Annual exceedance probabilities calculated as the inverse of RI. A sigmoidal regression was fit to the exceedance probability of the log-transformed discharge and 90% confidence intervals calculated about the model. Results of the model were used to establish annual exceedance probabilities of specific stormflow events.

A Partial Duration Series (PDS) record of floods on the Watauga River was calculated using instantaneous discharge collected at 15 minute intervals by the USGS gauge near Sugar Grove, NC for the period from 3 April 1986 to 26 March 2022. Centroid lag-to-peak time (TLPC) for the Watauga River Basin of 6.84 hours was calculated using a modified Snyder Equation (Equation 2) and multiplied by 3 for a conservative estimate of basin response time (Loukas and Quick, 1996).

$$t_l = C_b(LL_c/\sqrt{S_s})^{0.38} \tag{2}$$

t_l is lag time in hours, L is the stream distance of the longest flowpath in km, L_c is the stream distance from the basin centroid to the outlet in km, S_s is the dimensionless mean stream slope, and C_b is a coefficient assumed to be 0.42 from Loukas and Quick (1996). Watershed response time was used to identify clusters of peaks associated with a singular event and retain only the largest (Armstrong *et al.*, 2012). After simplifying peak clusters, a range of threshold discharge (TD) values were identified that resulted in an average of three to four events per year that met or exceeded that value (Langbein, 1949). The average value of 2100 cfs was selected as the TD for future calculations of low-magnitude flood trends, which returned 115 peaks above the TD from the present dataset (Figure 14). The 115 floods identified by PDS analysis were extracted from the full instantaneous discharge dataset and the number of floods recorded from each month was summed. A Monte Carlo simulation was performed to create 10,000 randomly distributed datasets containing 115 floods in 12 months to test the null hypothesis that the floods are randomly distributed throughout the year. The mean and standard deviation predicted number of floods were compared to the actual dataset returned by the PDS on a monthly basis using a Student's T-Test to identify significantly flood-rich and flood-poor months.

Abiotic Characterization

Modified Wolman Pebble Counts were conducted concurrently with as many macroinvertebrate collections as possible (Wolman, 1954). Each survey was established with the midpoint at the head of each study riffle, and 5 transects were established upstream and downstream with 10 m spacing. Beginning from the downstream end of the reach, 10 pebbles were selected from within the wetted width of the channel in a straight line working in a zig-zag pattern from one bank to the upstream end of the transect on the other bank (Bevenger and King, 1995). Along the straight lines from one bank to the other an effort was made to select one particle from each bank and the other 8 approximately equi-distant along the diagonal. Each individual particle was selected from under the toe of the right boot of the researcher with an averted gaze. Particles were measured across the intermediate axis with a gravelometer (Wildco part no. 3-14-D40, Yulee, FL) where applicable, otherwise measured across the shortest visible axis for buried boulders or classified as Bedrock, Sand, Silt, or Clay. One hundred particles were recorded for analysis of change through time. Where water depth and/or velocity prevented safe collection of a randomly selected particle the location was recorded as NaN. At each site D_{50}, Mean Grain Size, Sorting, Percent Bedrock, and Percent Fines were calculated from the Pebble Count Data (Supplemental File Site_statistics_pebbles.xlsx). Substrate metrics were averaged and compared across the three groups in the same way as benthic macroinvertebrate metrics.

Thalweg depth (cm) was measured at the head of each study riffle for each collection, along with current velocity (m/s) at the bed of the thalweg and at 40% of thalweg depth using a Global Water Flow probe. Water samples were collected for field measurement of turbidity (NTUs) at each collection. In-situ water chemistry at each collection was conducted using a YSI Xylem Multimeter recording temperature (˚C), pH (unitless), specific conductivity (μ S/cm), and dissolved oxygen (%). Current and Chemistry metrics were averaged and compared across the three groups in the same way as benthic invertebrate and substrate metrics except where mechanical issues in the field created data dropouts.

Macroinvertebrate Collection

Macroinvertebrate collections were performed in autumn 2020 and again in spring 2021 prior to dam removal to establish a baseline condition for each of the sites. A winter collection was obtained from the Tailrace for additional detail, but weather and resource constraints prevented a spring collection from the Impoundment. Post removal collections were scheduled for intervals of 1 week, 2 weeks, 1 month, 2 months, 3 months, 4 months, and 6 months; and were conducted as close to those intervals as weather and personnel availability would allow.

Riffle sites that could be safely waded were sampled using a modified NCDEQ Qual-4 method of a singe riffle-kick with the seine inspected for collected invertebrates for 30 labour-minutes, three D-net sweeps, three leaf packs, and a 30 minute visual survey (NCDEQ, 2016). Collected macroinvertebrates were preserved in the field using 80% EtOH and returned to the lab for identification to the Family taxonomic level (Merritt *et al.*, 2019; Morse *et al.*, 2017).

Before the dam removal, macroinvertebrates were collected from the Impoundment using only D-net sweeps and leaf debris collections. These non-standard methods were selected due to the depth of the water and absence of suitable riffle habitats for standard methods. The Qual-4 method described above was employed at this site as wadable conditions emerged following dam removal and natural excavation of sediments.

Each taxon was assigned a functional feeding group (FFG), functional habitat group (FHG), tolerance value (TV), and where literature was available, classified by voltinism life history

(Lenat, 1993; Merritt *et al.*, 2019; Morse *et al.*, 2017).

Macroinvertebrates were preserved in 80% EtOH by taxon and life stage where applicable in 1 dram archival vials for small taxa and 2.5 dram or 5 dram archival vials for large or numerous taxa and stored in Cornell Cabinets in the Appalachian State University Aquatic Ecotoxicology Lab Collections with Dr. Shea Tuberty.

An abundance matrix was created for every Family collected over the course of the study at each individual collection (Supplemental File Sitewise_abundance.csv). This matrix was used to calculate Nonmetric multidimensional scaling plots to determine separation between groups. For each collection the following were calculated: Total Abundance, Family Richness, Family IBI, Family Simpson, Family Shannon, Family Evenness, EPT Richness, Percent EPT, Ephemeroptera Richness, Percent Ephemeroptera, Plecoptera Richness, Percent Plecoptera, Trichoptera Richness, Percent Trichoptera, Percent Coleoptera, Percent Diptera, Percent Scraper, Percent Clinger, Percent Semivoltine (*K*-selected Taxa), Percent Multivoltine (*r*-selected Taxa), Percent Baetidae among Ephemeroptera, Percent Perlidae among Plecoptera, and Percent Hydropsychidae among Trichoptera (Supplemental File Site_statistics_bugs.xlsx). Voltinism data was obtained at various taxonomic levels from numerous sources, and where data was available at the genus or species level rather than family level the most common taxon from the collection was used as a proxy (Beaty, 2015; Merritt *et al.*, 2019). Where voltinism information was unavailable the taxon was assigned NA for the life history and calculations excluded that taxon. Beta Diversity was calculated for each time step using the Upstream Reference, Dam Impacted, and Downstream Impacted groups by averaging all sites from each group for each time step. Significance was determined using one-way ANOVA and Tukey Post-Hoc tests.

Abiotic Drivers of Macroinvertebrate Response

Linear and multiple linear regressions were used to identify which habitat parameters were important for describing responses of benthic macroinvertebrate community metrics to the dam removal. For linear regressions at each of the three study groups, all of the measured habitat parameters were used individually as predictor variables against all of the calculated benthic invertebrate community metrics individually as response variables. For multiple linear regressions the same method was utilized as linear regressions excepting that predictor variables were combined where they related to each other. D_{50} and Percent Fines were utilized together as substrate variables to identify any community response they might predict. Bed Velocity and 40% Depth Velocity were utilized together as flow variables to identify any community response they might predict.

Habitat vectors were applied to NMDS plots to determine influence on dissimilarity among sites. Principal component analysis (PCA) was attempted, however data dropouts in many of the habitat parameters prevented successful execution.

Results

Hydrology and Geomorphology

The Ward Mill Dam was removed on 16 May 2021, during the falling limb of the hydrograph (Figure 17). Since that date only one major flood has occurred on the Watauga River, when Tropical Storm Fred produced 15 cm of rain as measured in the nearby town of Boone and the Watauga River peaked in the early hours of 18 August 2021 at 6550 cfs. This event magnitude has a recurrence interval of 2.4 years, and an annual exceedance probability of 0.43 ± 0.08 (Figure 18). Following this event, qualitative observation of the river in the former impoundment and downstream showed significant changes visually. Within the impoundment the rapids became more defined, islands disappeared, and pools formed. Downstream, the tailrace re-established a thalweg near to the original location and large sandbars appeared 1 km downstream. The next largest event occurred on 5 February 2022 when the Watauga River peaked at 1640 cfs, a flow less than the 1 year flood magnitude. A round of macroinvertebrate and sediment surveys was planned for approximately a week after that event but high water and low temperatures prevented field work at that time. The minor high-water event of 1250 cfs on 2 July 2021 caused some changes to the impoundment but continuous erosion continued after the peak had passed.

Seasonal flood analysis for the Watauga River indicated that April and November are flood-rich (p = 0.009, p = 0.025), and June is flood-poor (p = 0.011) (Figure 19). No other month differed significantly from the simulated number of floods.

Following dam removal the cobble and boulder beds within the riffles of the tailrace had been replaced with sand and gravel, burying the original bed under more than a meter of fine sediment (Figure 15). Few cobbles remained on the surface, and where present they were fully embedded with no interstitial habitat beneath them. Fluctuations in the riverbed as the channel attempted to re-establish itself cut off overhanging vegetation and root mats from aquatic organisms. Within the former Impoundment the bed was composed of shifting sands and gravels, with frequent collapses of sandy banks (Figure 16). Occasional boulders and cobbles emerged from the stored sediments, but were fully embedded with no interstitial habitat beneath them as in the Tailrace. Incision of the riverbed into stored sediment detached overhanging vegetation from the water as muddy and sandy banks increased in height. Leaf packs are heavily seasonal in the Watauga River, not persisting in abundance through the early study period. No structure existed in the Impoundment or Tailrace to capture and accumulate the few allochthonous leaves into leaf pack habitats.

High natural variability within pebble counts between sites and at repeat surveys of the same site widened confidence intervals such that the only statistically significant difference in the median grain size (D_{50}) comparisons is that between the Upstream Reference group and the Dam Impacted group one week post removal (Figure 20). Across the upstream sites D_{50} fluctuates about a stable value indicating natural variability. Dam Impacted sites did not have pebble counts conducted prior to removal, however following removal D_{50} remained around 20 mm intermediate diameter, low compared to Upstream Reference sites and Downstream Impacted sites throughout the entire study period. Downstream Impacted sites appear to show an initial decrease in D_{50} with rapid recovery, however variability within this group is extreme (Figure 21). The Pasture Site, located 1 river km downstream of the dam showed a steep decrease in D_{50} immediately following the dam removal, and another decrease immediately following the flood. Large sandbars appeared at the Pasture Site following the flood, burying the already embedded substrate at this site under additional fine sediment. The Hubert Thomas Rd. bridge site experienced an unexpected and unexplained spike in D_{50} at 4 months post removal. The extremity of this data point increases the mean of the group above the mean value of the other two sites within the group and confounds the Downstream Impacted recovery trajectory. Below freezing air temperatures and deep water for the 6 month survey prevented pebble counts at Camp Broadstone, Rominger Rd. bridge, and US Hwy. 321 bridge. These data dropouts at historically high D_{50} sites cause the 6 month D_{50} averages to be dominated by historically low D_{50} sites where pebble counts were possible and does not reflect trends at the Upstream Reference or Downstream Impacted sites. There were no statistically significant differences between any of the three groups for Percent Fines at any time step.

Benthic Macroinvertebrates

The initial collections from the tailrace of the dam and the newly exposed impoundment struggled to find any organisms at all except on rare driftwood. The first collection in the Tailrace only located four insect individuals across four Families: one Perlid, one Baetid, one Elmid, and one Chironomid; along with two Pleurocerid snails (genus *Leptoxis*) and two Oligochaetes. The first collection in the Impoundment contained more organisms, but was conducted two days later and contained more driftwood where the organisms were almost exclusively found.

NMDS plots show much greater spread among Tailrace and Impoundment sites than all other sites (Figure 22). The large spread among the two Dam Sites, along with the overlap of the Pasture Site and Hubert Thomas Rd. bridge site with the upstream reference sites indicated that the biological impacts of the dam removal did not spread as far downstream as anticipated.

Reanalysis of the NMDS plots with the sites grouped into Upstream Reference, Dam Impacted, and Downstream Impacted showed a clear separation between the Dam sites and both other groups (Figure 23). All subsequent analyses were conducted using the three groups identified by NMDS plots rather than the two initially hypothesized groups of Reference and Impacted for increased resolution of results.

Impacts at the Dam Sites began immediately following the removal, with significant differences found in various metrics relative to Upstream Reference Sites (Figure 24). During the first survey significant reductions were found in Abundance, Family level Richness, both Diversity metrics, Trichoptera Richness, and EPT Richness at the Dam sites (Figures 25 to 30). The Clinger Functional Habitat Group (FHG) showed a decrease in relative abundance at the Dam site following removal, while *r*-selected taxa (multivoltine life history) and Percent Diptera showed increases (Figures 31 to 33). The Scraper Functional Feeding Group (FFG) showed a decrease in relative abundance at the Dam sites, but was delayed until the 2 week post dam removal survey period (Figure 34). The same delayed response period showed an increase in percent Ephemeroptera at the Dam sites (Figure 35). Abundance of benthic invetebrates recovered quickly at the Dam sites, and within one month of the removal there was no significant difference between any of the three groups. Both diversity metrics recovered to meet the Upstream and Downstream groups within 4 months following removal, along with Trichoptera Richness and EPT Richness. Within 6 months following the removal all biological metrics except Family Richness had recovered to no statistical difference between the three sites.

The Downstream Impacted Sites did not show any statistically significant changes to benthic invertebrate community composition until the second week of surveys (Figure 36). At the second week of surveys Trichoptera Richness and EPT Richness were significantly lower at the Downstream Impacted sites compared to Upstream Reference sites. Those impacts only persisted for a single survey; one month post removal they showed no statistical depression relative to upstream sites. Also, the Scraper FFG was significantly lower at the two month survey for the Downstream Impacted sites, but had recovered by the three month survey.

Benthic macroinvertebrate communities at the Downstream Impacted sites were not affected by the dam removal to the extent hypothesized before the project began. Trichoptera richness was significantly different from the Upstream Reference sites two weeks post removal, however this is due to an increase in Trichoptera families upstream, rather than a decrease downstream. The EPT richness follows the same pattern of significance, and is driven primarily by the increased upstream Trichoptera richness. Following the deviation at 2 weeks post removal the Downstream sites are not significantly different from the Upstream sites in Trichoptera or EPT richness. Scrapers at the Downstream Impacted sites were significantly lower than at the Upstream Reference sites two months following removal. The Pasture Site showed a large increase in scrapers immediately following the dam removal, then a decline lasting two months (Figure 37). The Hubert Thomas Rd. bridge site showed a similar pattern, but less pronounced. This is the same pattern seen in the Dam Impacted sites. However the US Hwy. 321 bridge did not show this pattern, instead following the same pattern of a gradual increase in scrapers for three months following the removal seen at the Upstream Reference sites.

Invertebrate Response to Habitat Change

Regression analysis of benthic macroinvertebrate response did not result in any relationships which were well predicted by substrate parameters. Using a lower limit R^2 value of 0.4, Bed Velocity and 40% Depth Velocity were individually able to predict Percent Plecoptera, Trichoptera Richness, and Percent Semivoltine at the Dam sites (Figures 38 to 40). Multiple linear regressions using the same lower limit R^2 value of 0.4 and comparing response variables to both velocity measurements were able to predict Abundance, Family Richness, EPT Richness, Percent EPT, Ephemeroptera Richness, Percent Plecoptera, Trichoptera Richness, Percent Semivoltine, and Percent Hydropsychidae among Trichoptera at the Dam sites (Table 2). No habitat parameters were able to predict benthic invertebrate community metrics indivdually or when paired at either the Upstream Reference sites or at the Downstream Impacted sites. No habitat vectors were significant ($p > 0.05$) when applied to the NMDS plots of Family abundance.

Discussion

Dams occur over a broad range of stream orders, basin size, and slope (Bellmore *et al.*, 2017; Foley *et al.*, 2017b). Studies of other dam removals on rivers in these other geographic settings indicate that the recovery timeline is variable, with some dam removals showing little to no impact to benthic macroinvertebrates while other removals depress community metrics for years (Gillette *et al.*, 2016; Orr *et al.*, 2008). With the high variability of geographic settings for dam removals and subsequent impacts, predictions about dam removal impacts on downstream channel morphology and aquatic organisms can be difficult. Some general patterns are emerging from dam removal research of downstream aggradation, bed fining, habitat homogenization, and decreases in benthic invertebrate community indexes. It may be generalizable to other systems that a steep, cobble-bed river like the Watauga will have a longer recovery period, as the impoundment was filled with coarse material in the sand-gravel range with some cobbles above the buried channel (East *et al.*, 2018). However, few removals in moderate to high gradient basins have been performed, and few studies have been conducted in those basins (Foley *et al.*, 2017b).

The reduction of numerous benthic macroinvertebrate community metrics at the Downstream and Dam Impacted sites and subsequent recovery within six months to Upstream Reference condition is consistent with studies of dam removals in other geographic, climatic, and elevation provinces (Chiu *et al.*, 2013; Thomson *et al.*, 2005). Family level taxonomic richness remained low at the Dam Impacted sites at six months post-removal, however the recovery of community metrics such as percent EPT and percent Scraper indicate that all of the ecological roles are filled, but with less diversity.

The Watauga River was in the process-driven phase of recovery as described by Pearson *et al.* (2011) until the 18 August 2021 event, which hastened the transition to the event-driven phase. Flood seasonality modeling indicated that August is not a particularly flood-rich month, and that a flood should not have been expected to occur until November. Future dam removal planners may use the method described in this paper to plan dam removal projects to coincide with the end of a flood-rich period and the beginning of a flood-poor period. This planning would allow the maximum amount of time for the process-driven phase of sediment excavation to continue uninterrupted before either the event-driven phase is naturally reached or seasonal floods return and precipitate a transition.

Cross sections and longitudinal profiles were surveyed by Josh Platt of Appalachian State University (2022), but were not conducted as part of this research. Qualitative changes to the channel were visible even during low flows prior to the flood. Incision was noted in the impoundment between successive visits to field sites, as the rapids became increasingly distinct. Channel widening was notable, as bank failures happened frequently enough that they were a safety concern during macroinvertebrate collections and pebble counts. Following the flood, the channel had changed considerably as described above. However limited geomorphic work was done by further low flows, or even successive marginally high water events, after redistribution of sediment by the large flood. The early arrival of a 2 year flood hastened the transition from the process-driven phase to the event driven phase (Harrison *et al.*, 2018).

This hastened change to event-driven recovery suggests that a flood event early in the recovery can overshadow process-driven recovery, and rapidly excavate the impoundment. However, persistent aggradation and fining downstream urges caution that such an early transition may make it difficult for downstream reaches to incise back to the pre-removal channel without further high flow events, prolonging downstream recovery. Fine sediment deposited by the flood has developed a layer of coarser armor composed of fine gravel to small cobbles, however boots sink through this layer revealing sand beneath. Further high flow events will be required to erode the armored material remaining on the downstream riverbed.

Benthic macroinvertebrate communities at the Dam Impacted sites were significantly impaired following the dam removal when compared to Upstream Reference sites, but the temporal extent of the impact was shorter than predicted. The only community metric which showed significant differences at the 6 month survey was Family level taxonomic richness. The decrease of multivoltine taxa at 4 months post removal indicates that *r*-selected taxa no longer dominate the community and that *K*-selected taxa have recolonized the disturbed sites. The recovery of both diversity metrics at 3 months to the Upstream Reference sites shows that recolonization from upstream and in-situ reproduction has reduced the dominance of opportunist Families which dominated the community immediately following disturbance.

Changes to habitat parameters were not as insightful as expected for predicting benthic invertebrate response. At no point during the study did Percent Fines differ at the Dam or Downstream Impacted sites from the Upstream Reference sites. D_{50} remained significantly lower at the Dam and Impacted sites relative to Upstream Reference sites for the entirety of the study period. Neither habitat parameter alone or in conjuction were able to predict more than 40% of the variablility in any benthic invertebrate community metric. This suggests that even where large areas of habitat have been buried by sand and gravel, enough cobbles remain in the thalweg and other fast areas of flow to support a benthic community in pockets of suitable habitat.

Dam removal studies within high gradient, coolwater, montane streams have been limited (Bellmore *et al.*, 2017; Foley *et al.*, 2017a,b). Studies conducted on small dam removals in the Cascade Mountains of Oregon, USA found recovery of benthic invertebrates downstream of the dam removal to upstream reference condition within one year following the removal (Tullos *et al.*, 2014). The removal of a small dam in tropical Taiwan showed a similar response with downstream benthic invertebrate abundance recovering to match upstream reference within 4 months, however much of the abundance was due to fast reproducing *r*-selected taxa (Chiu *et al.*, 2013). A study of a small dam on a coldwater stream in Wisconsin, USA found an immediate decrease in benthic invertebrate abundance which recovered but persisted through the one year study; and was more pronounced for sensitive taxa such as Ephemeroptera and Trichoptera relative to Diptera (Orr *et al.*, 2008). The varied geography and findings of previous macroinvertebrate studies highlights the fundamental difficulty of predicting dam removal response, that these are natural systems that respond to a multitude of unpredictable variables at various spatial scales (Carlson *et al.*, 2018; Pollard and Reed, 2004). This study conducted in the Appalachian Mountains of North Carolina, USA most closely resembles the findings of Tullos (2014) in another temperate mountainous environment with benthic invertebrate recovery within one year following the dam removal. This suggests that in montane systems the benthic macroinvertebrate recovery following a small dam removal may be rapid and occur before geomorphic recovery, however further research will be required to verify this (Foley *et al.*, 2017a; Tullos *et al.*, 2014).

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Tables

Table 1: *Locations and relative distances of the study sites. River kilometers refers to the distance of sites downstream of Ward Mill Dam.*

Site Name	River Kilometers	Latitude	Longitude	Elevation (m ASL)	Grouping
Camp Broadstone	-12.5	36.1938	-81.7572	820	Upstream Reference
NC Hwy. 194 bridge	-7.8	36.2168	-81.7863	809	Upstream Reference
Rominger Rd. bridge	-0.8	36.2387	-81.8235	797	Upstream Reference
Ward Mill Impoundment	-0.1	36.2407	-81.8296	794	Dam Impacted
Ward Mill Tailrace	0.1	36.2420	-81.8311	791	Dam Impacted
Pasture Site	1	36.2473	-81.8310	789	Downstream Impacted
Hubert Thomas Rd. bridge	2	36.2512	-81.8414	787	Downstream Impacted
US Hwy. 321 bridge	3.4	36.2596	-81.8601	785	Downstream Impaced

Table 2: *Multiple regression models with two or more related continuous predictor variables are not conducive to visualization in 2 dimensions. Coeffecients of model results with $R^2 \geq 0.4$ are presented here in a tabular fashion. Significant relationships ($p \leq 0.05$) are ephasized. The Bed Velocity and 40% Depth Velocity columns indicate the coefficients for those variables in the multivariate regression equation.*

Response	R^2	*p* value	Intercept	Bed Velocity (m/s)	40% Depth Velocity (m/s)
Abundance	0.490	0.133	191.9	249.1	-209.4
Family Richness	0.501	0.124	20.8	20.1	-16.6
EPT Richness	0.745	**0.017**	15.5	15.6	-14.9
Percent EPT	0.700	**0.027**	102.8	80.1	-69.2
Ephemeroptera Richness	0.824	**0.005**	7.6	9.1	-7.9
Percent Plecoptera	0.447	0.169	3.7	18.7	-1.8
Trichoptera Richness	0.646	**0.044**	5.3	3.5	-4.9
Percent Semivoltine	0.469	0.150	-3.0	-3.4	15.3
Percent Hydropsychidae	0.877	**0.002**	111.0	240.9	-156.6

Figures

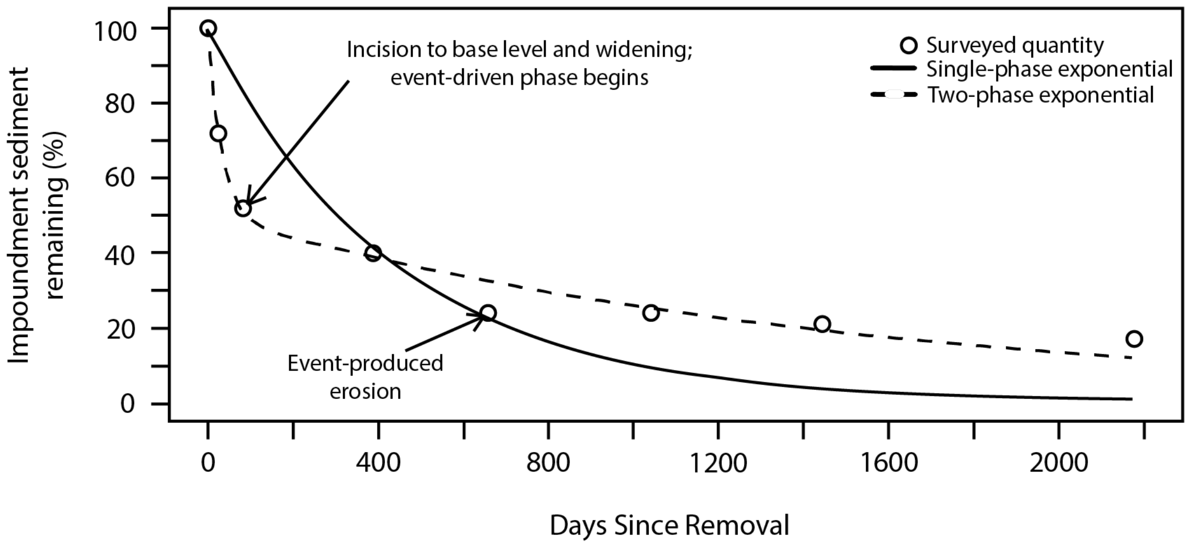

Figure 1: *Surveyed sediment remaining in the Merrimack Village Dam reservoir following removal (Pearson* et al.*, 2011). Later four surveys show event-driven incision producing negative residuals to two-phase model initially, rising to a positive residual between high flow events. Figure modified from Collins* et al. *(2017).*

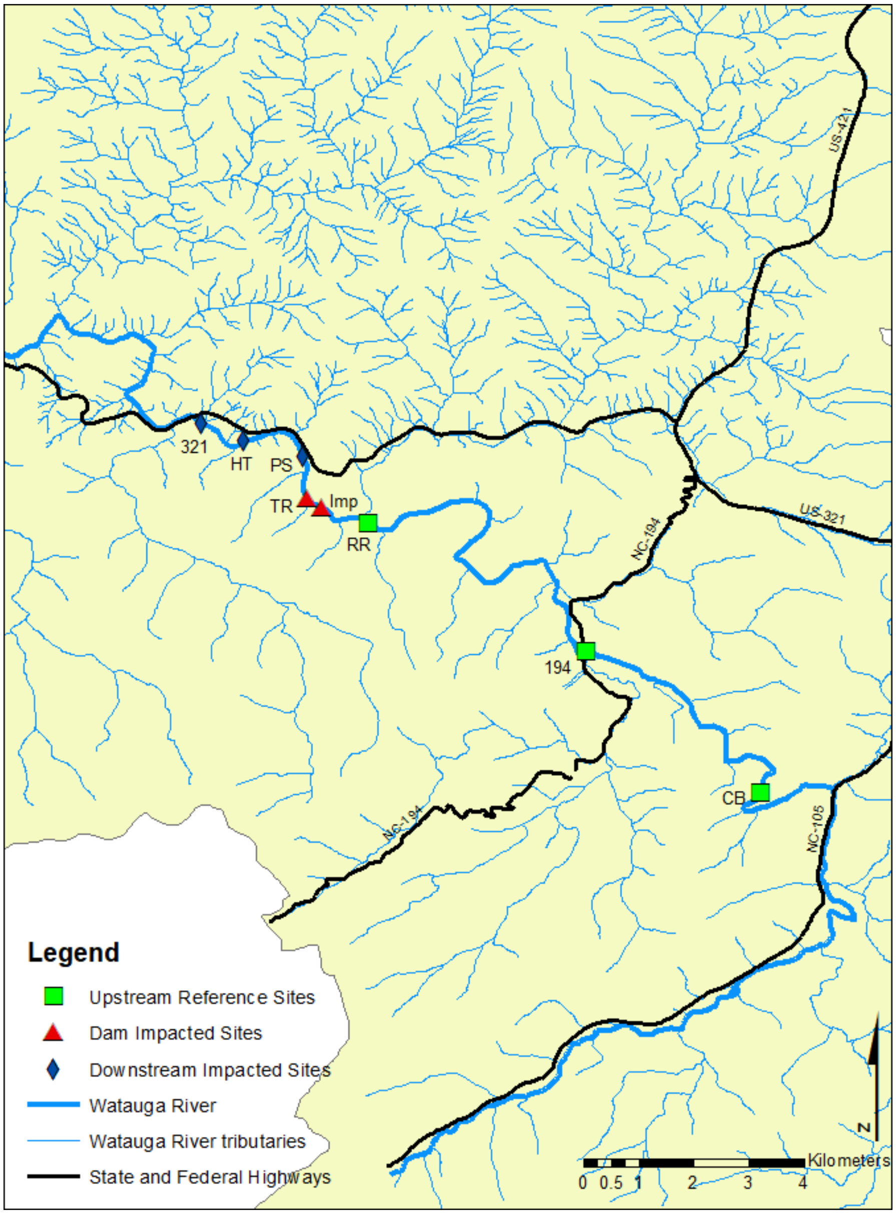

Figure 2: *Relative locations of collection sites along the Watauga River. All sites are located on the mainstem of the Watauga River, which flows from southeast to northwest within the study area. From upstream, the site order is Camp Broadstone (CB), NC Hwy. 194 bridge (194), Rominger Rd. bridge (RR), the Ward Mill Impoundment (Imp), the Ward Mill Tailrace (TR), the Pasture Site (PS), Hubert Thomas Rd. bridge (HT), and US Hwy. 321 bridge (321). The former Ward Mill Dam site is situated between the Ward Mill Impoundment and Ward Mill Tailrace. Streams mapped by the USGS, mapping resolution inconsistent within the basin.*

Figure 3: *Satellite imagery of the site at Camp Broadstone with the study riffle marked. Flow direction is from the bottom left to top right. Pebble counts were conducted in a section extending 50 m upstream and downstream of the mark.*

Figure 4: *Satellite imagery of the site at the NC Highway 194 bridge with the study riffle marked. Flow direction is from right to left. Pebble counts were conducted in a section extending 50 m upstream and downstream of the mark.*

Figure 5: *Satellite imagery of the site upstream of the Rominger Road bridge with the study riffle marked. Flow direction is from the bottom right to top left. Cove Creek enters the Watauga River at the top, and the USGS Watauga River gauge is approximately halfway between the confluence and the bridge. Pebble counts were conducted in a section extending 50 m upstream and downstream of the mark.*

Figure 6: *Satellite imagery of the Ward Mill Impooundment prior to dam removal. The approximate location where the study riffle emerged is marked. Flow direction is from the bottom right to top left.*

Figure 7: *Image of the dam removal as seen from upstream showing the mud and sand bed as it is exposed by receeding water within the former impoundment. Image credit to Josh Platt, 2021.*

Figure 8: *Satellite imagery of the Ward Mill Tailrace prior to dam removal with the study riffle marked. Flow direction is from the bottom right to top left. Pebble counts were conducted in a section extending 50 m upstream and downstream of the mark.*

Figure 9: *Image of the dam during removal showing downstream deposition of sediment burying the existing channel. Image credit to Josh Platt, 2021.*

Figure 10: *Satellite imagery of the Pasture Site with the study riffle marked. Flow direction is from the bottom right to top left. Pebble counts were conducted in a section extending 50 m upstream and downstream of the mark.*

Figure 11: *The Pasture Site as seen from the banks showing sandbars deposited by the flood on 18 August 2021. Image credit to Josh Platt, 2021.*

Figure 12: *Satellite imagery of the site at the Hubert Thomas Road bridge with the study riffle marked. Flow direction is from right to left. Pebble counts were conducted in a section extending 100 m downstream of the mark.*

Figure 13: *Satellite imagery of the site at the US Highway 321 bridge with the study riffle marked. Flow direction is from the bottom right to top left. Pebble counts were conducted in a section extending 50 m upstream and downstream of the mark.*

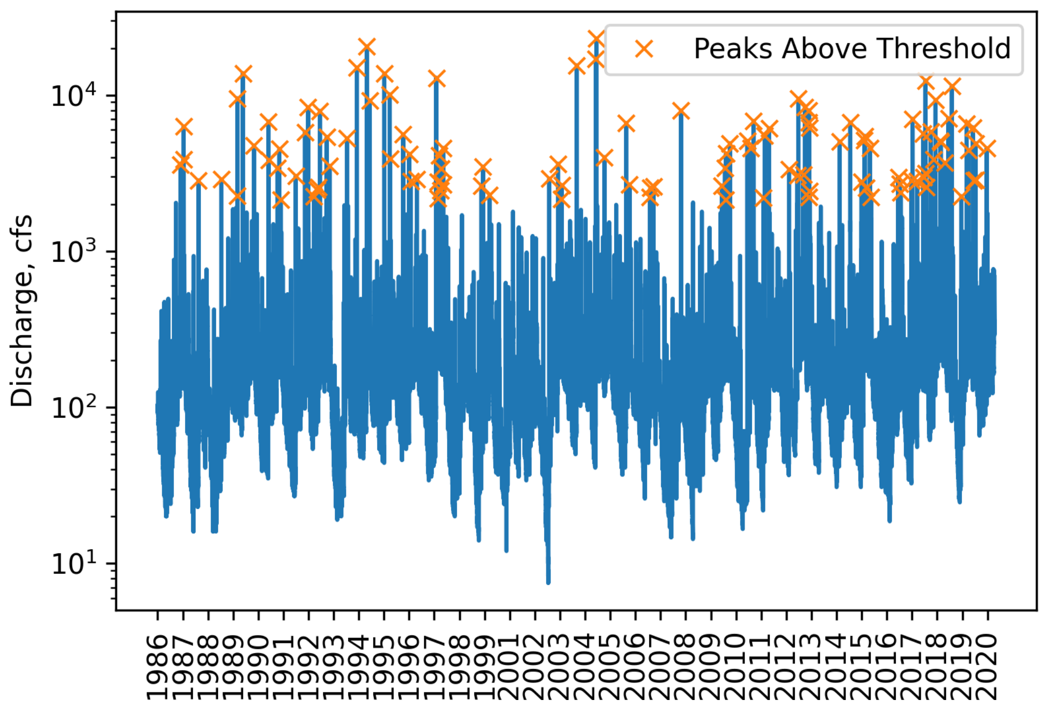

Figure 14: *Flood peaks identified in instantaneous gauge data between 3 April 1986 and 26 March 2022. Threshold discharge set at 2100 cfs, 115 peaks identified in the time series.*

Figure 15: *Image of the island downstream of the former dam site showing deposition of sand and gravel above the original bed. Image credit to Josh Platt, 2021.*

Figure 16: *Image of the former impoundment showing bed coarsening from mud banks to a sand and gravel bed with isolated boulders. Image credit to Josh Platt, 2021.*

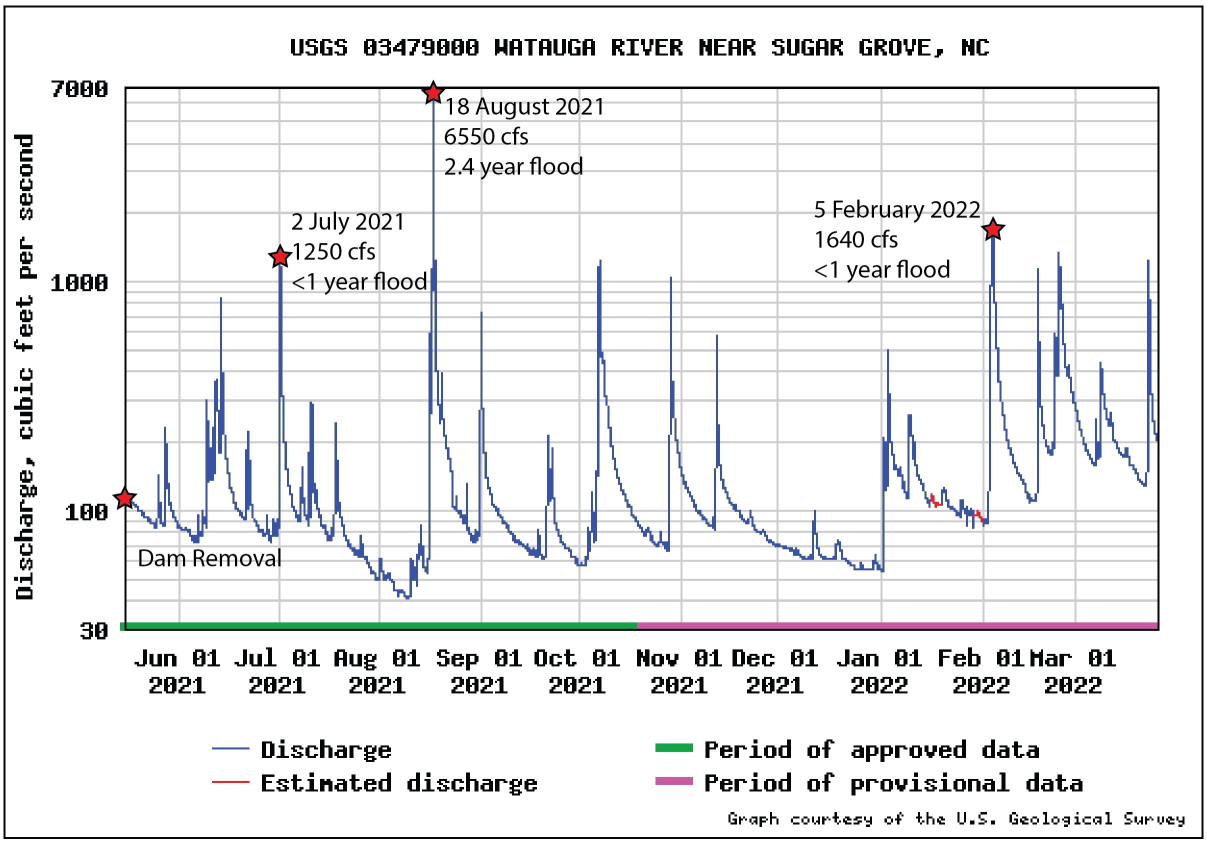

Figure 17: *Hydrograph of the Watauga River at the Rominger Rd. bridge, 0.89 km upstream of the dam and downstream of any major tributaries before the dam. Time series displayed begins on 16 May 2021, the day of removal, and proceeds to 26 March 2022.*

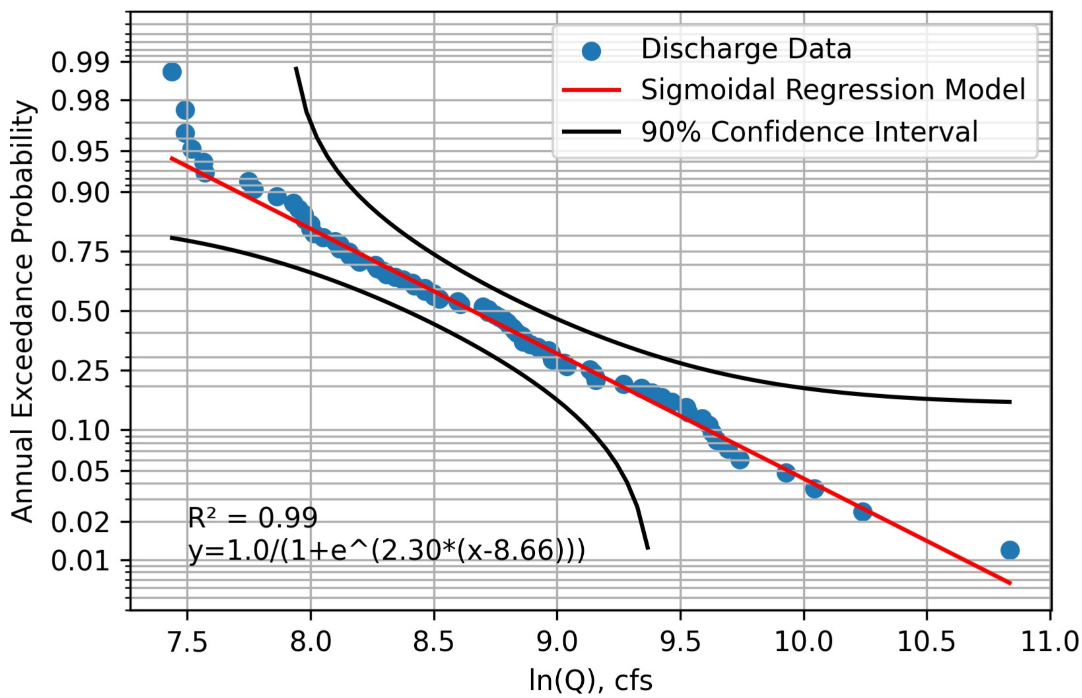

Figure 18: *Flood frequency model for the Watauga River near Sugar Grove, NC for the time series 1916, 1940-2022.*

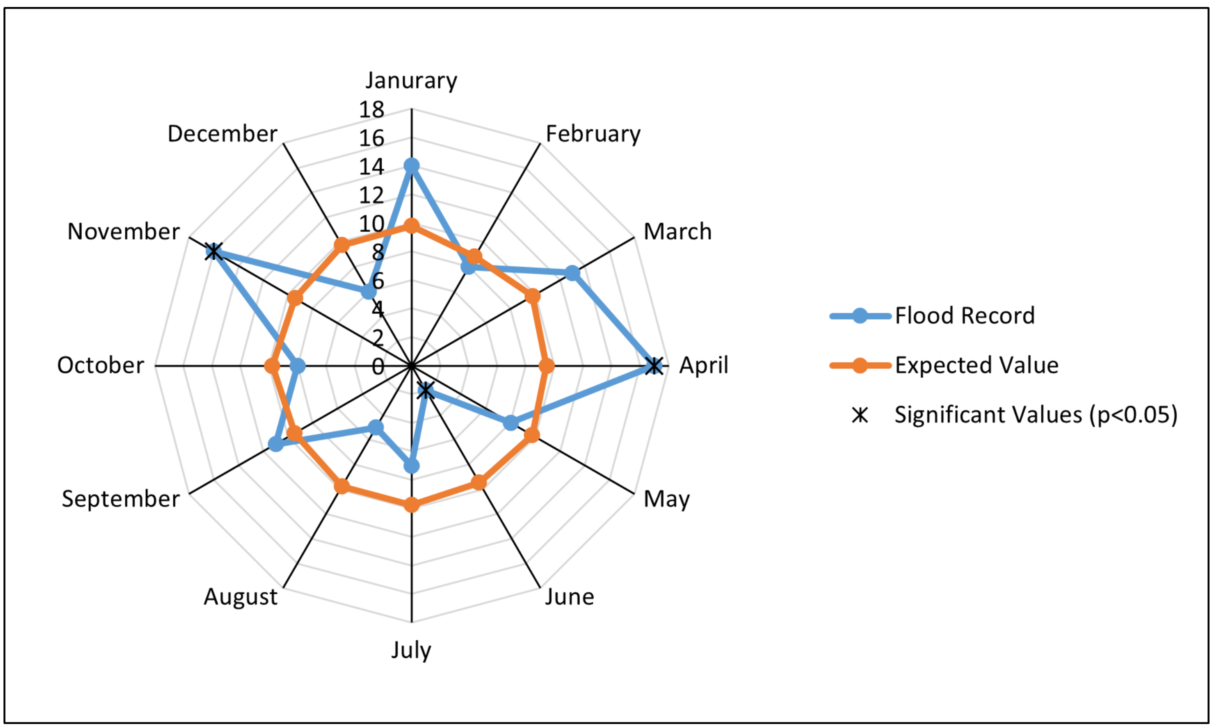

Figure 19: *Predicted and actual number of floods per month for the period of 1986-2021. Significance established through Monte Carlo simulation of randomly distributed floods of n=115.*

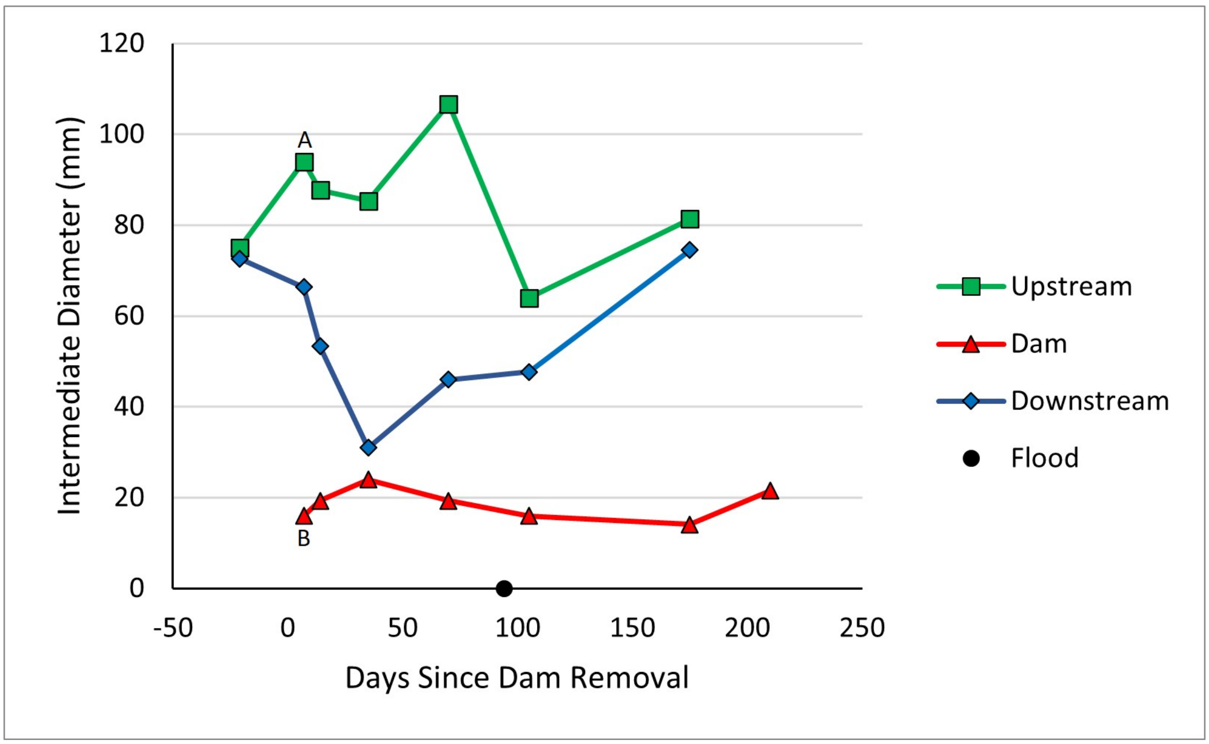

Figure 20: *Median grain size (D_{50}) for the three groups. One week post removal the Dam site differs from Upstream, but neither differs from Downstream. Data dropouts at 6 months for Upstream and Downstream result from incomplete pebble counts at Camp Broadstone, the Rominger Rd. bridge, and the US Hwy. 321 bridge due to low temperatures and high water.*

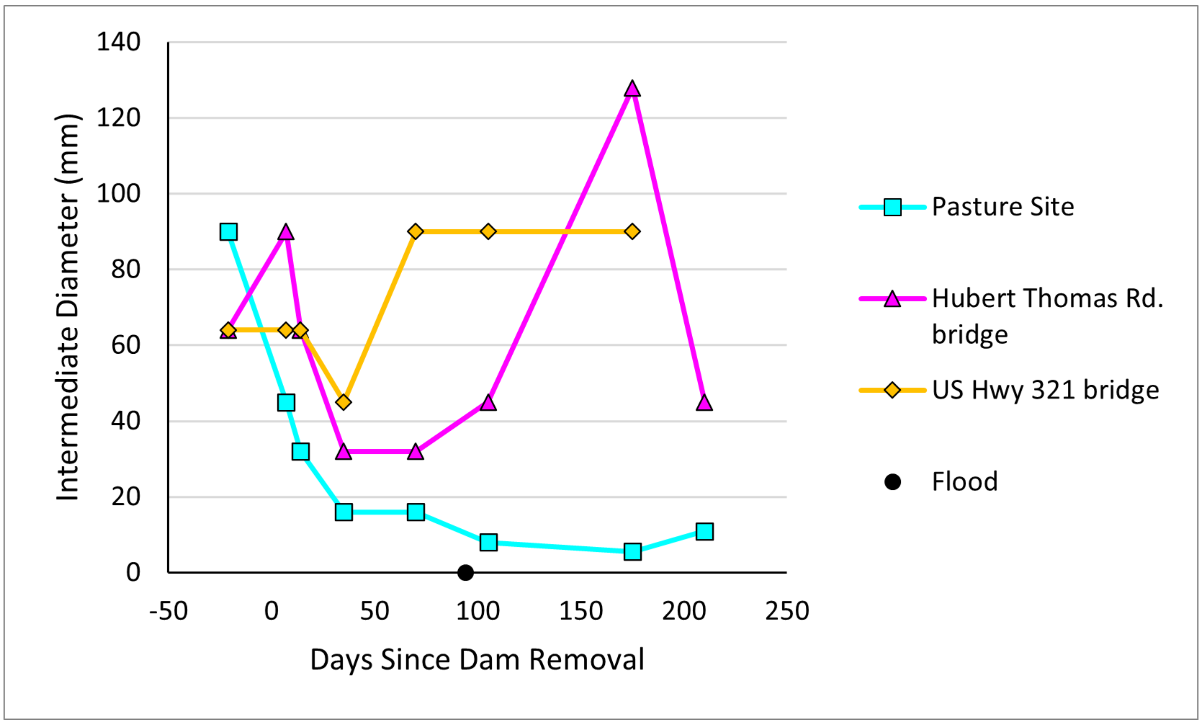

Figure 21: *Median grain size (D_{50}) for the three sites in the Downstream group. Data dropout at 6 months for the US Hwy. 321 bridge results from an incomplete pebble count due to low temperature and high water. The increased D_{50} at 4 months for Hubert Thomas is unexplained, no geomorphological difference was noted in the field.*

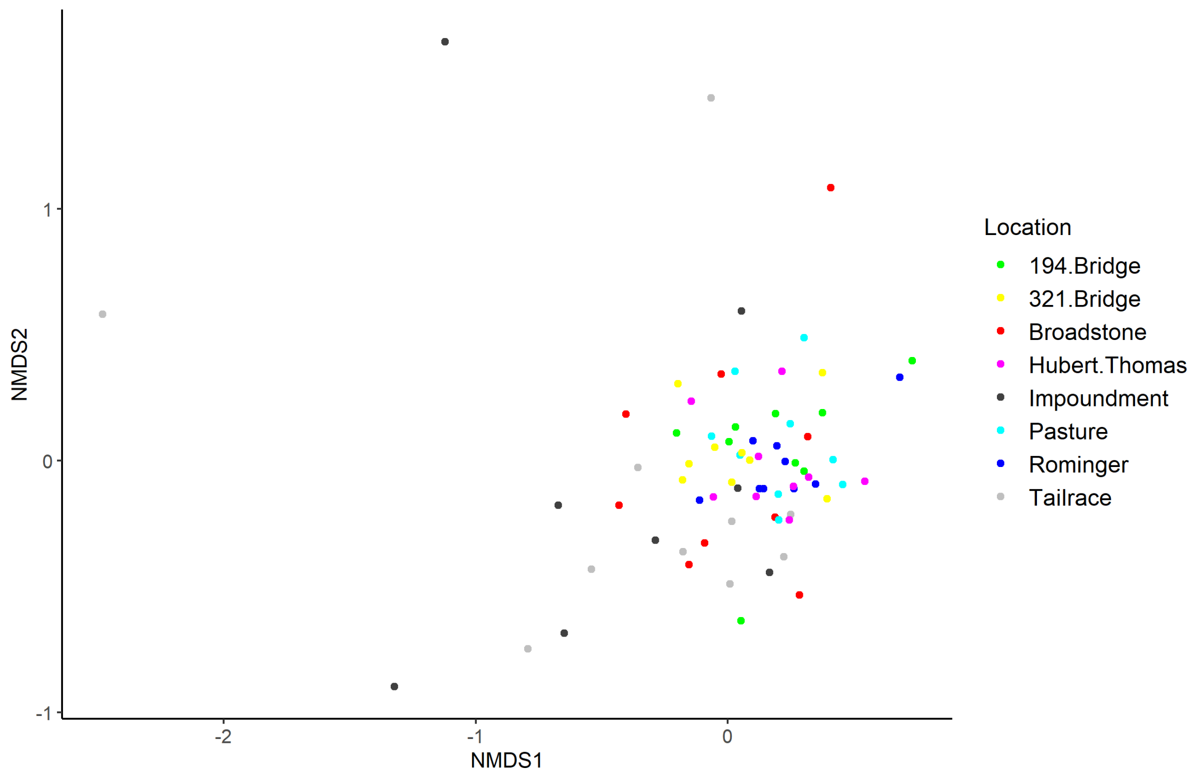

Figure 22: *NMDS using a Family level abundance matrix for all collections. Each site is individually colored. Stress was <0.2. The Pasture Site and Hubert Thomas are within the spread of Broadstone, 194, and Rominger. The Impoundment and the Tailrace partially overlap with other sites but have a much larger spread. This resulted in changing the study design from Reference/Impacted to Reference/Impacted/Impacted in order to increase resolution.*

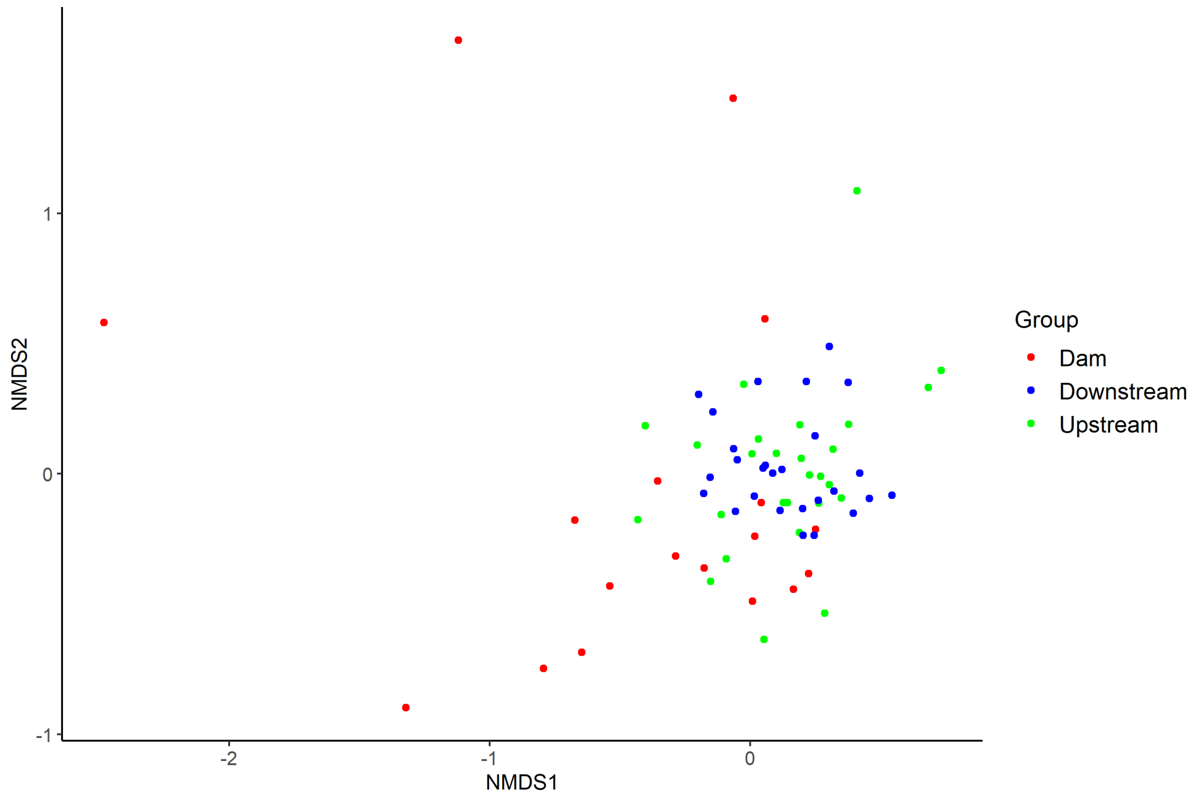

Figure 23: *NMDS using a Family level abundance matrix for all collections. Sites have been grouped into Upstream Reference, Dam Impacted, and Downstream Impacted. Stress was <0.2. The Downstream Impacted are within the spread of the Upstream Reference. The Dam Impacted partially overlap with other groups but have a much larger spread. This confirmed changing the study design from Reference/Impacted to Reference/Impacted/Impacted in order to increase resolution.*

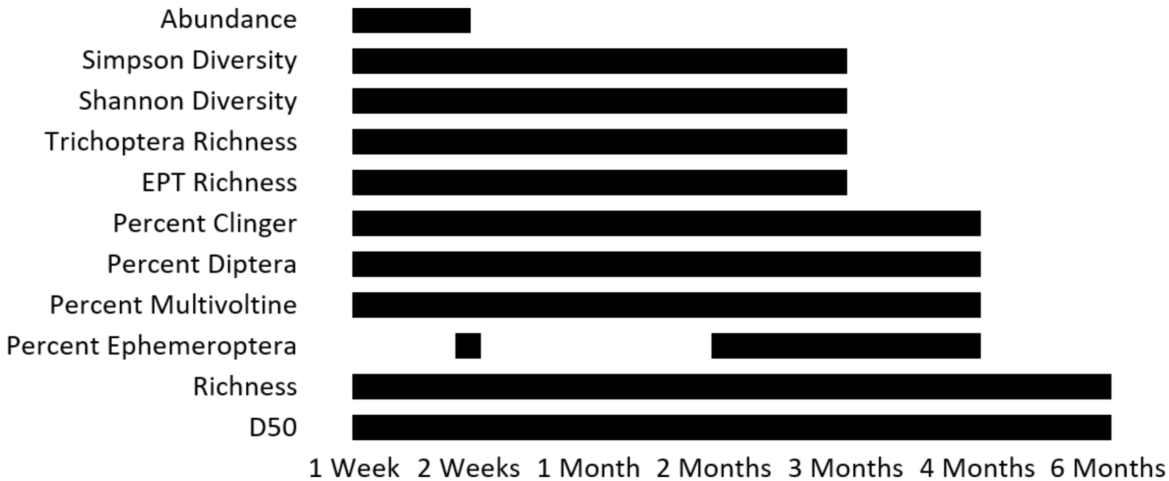

Figure 24: *Statistically and ecologically significant metrics at the Dam Impacted sites as compared to Upstream Reference sites. Black horizontal bars indicate the duration of significance. Abundance was significantly decreased at the 1 Week and 2 Week surveys, not significantly different at 1 Month, and significantly decreased at 6 Months. Percent Ephemeroptera was significantly increased at 2 Weeks, not significantly different at 1 Month, and significantly increased at 2 Months.*

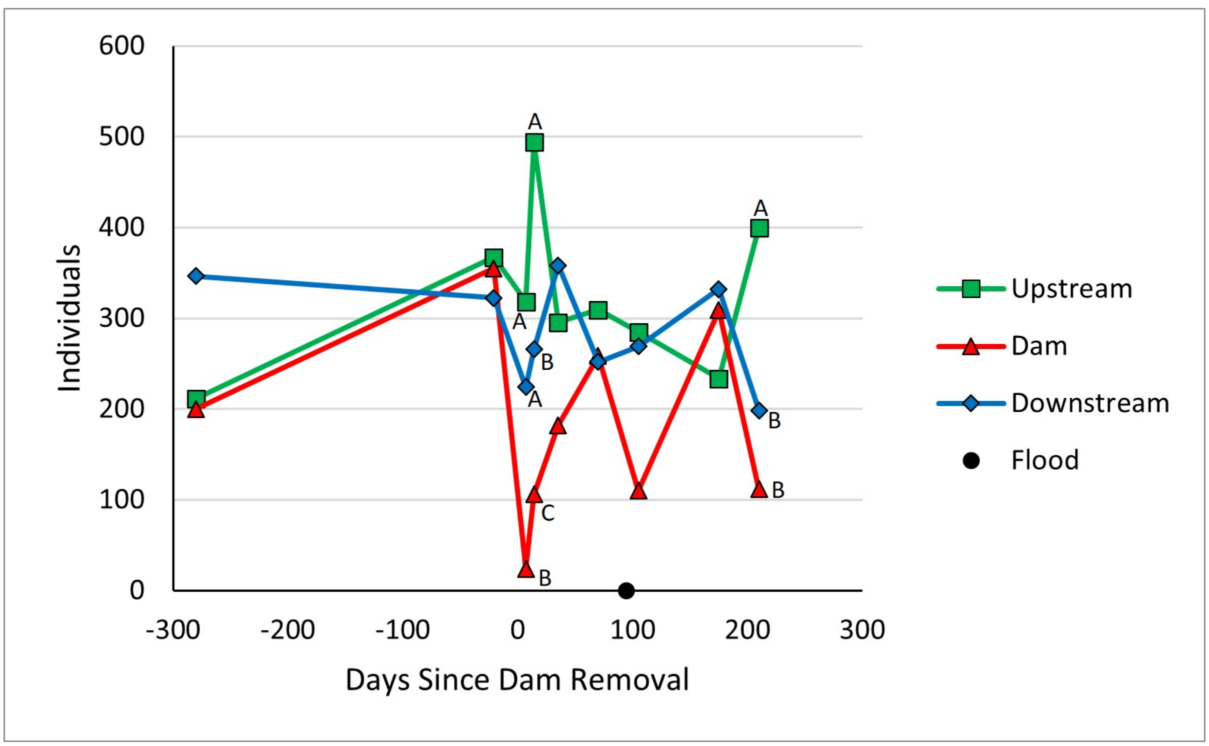

Figure 25: *Benthic macroinvertebrate abundance for the three groups. One week post removal the Dam differs from both Upstream and Downstream, but they do not differ from each other. Two weeks post removal all three groups differ from each other. Six months post removal Upstream differs from both the Dam and Downstream, but they do not differ from each other.*

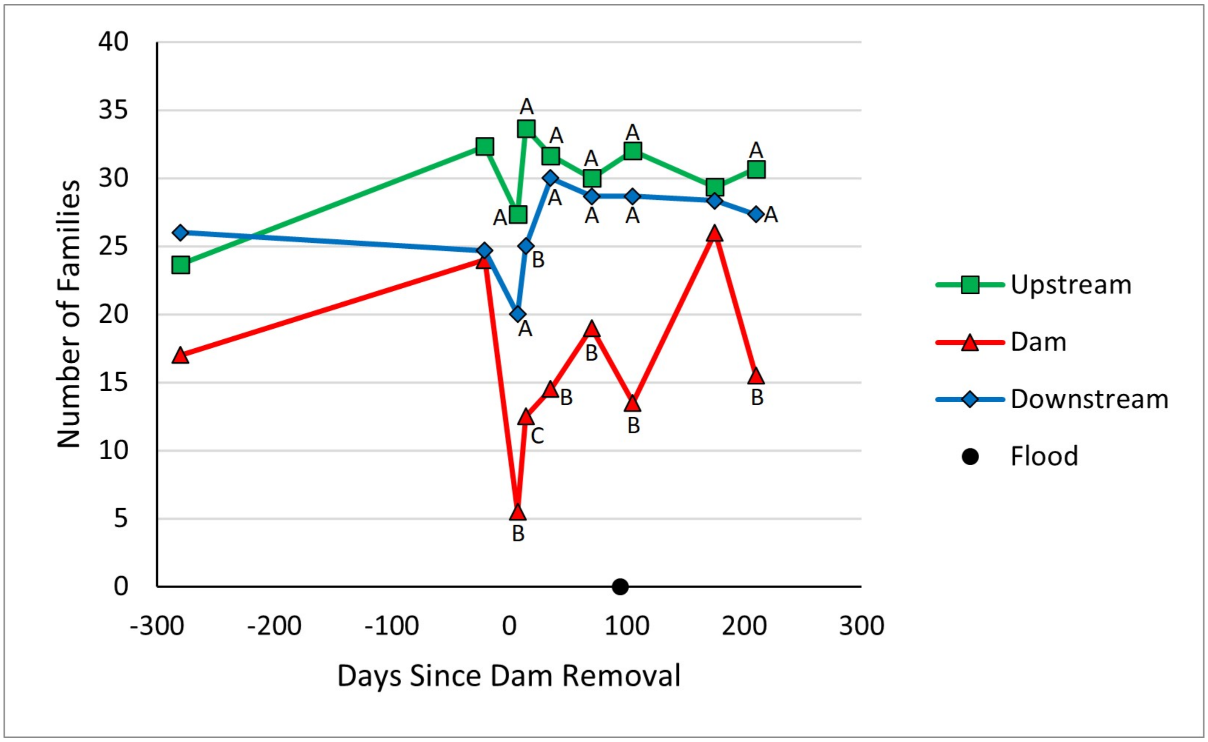

Figure 26: *Benthic macroinvertebrate richness at the Family taxonomic rank for the three groups. One week post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two weeks post removal all three groups differ from each other. One month post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Six months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

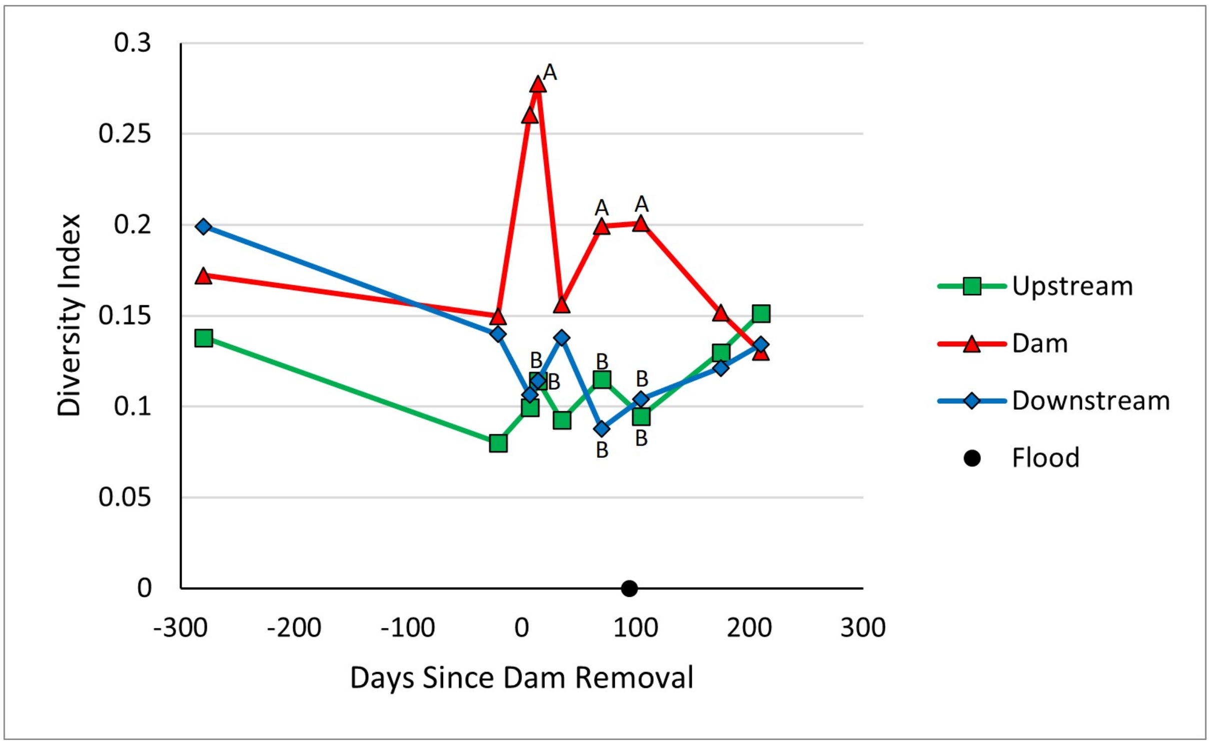

Figure 27: *Simpson diversity at the Family taxonomic rank for the three groups. Two weeks post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

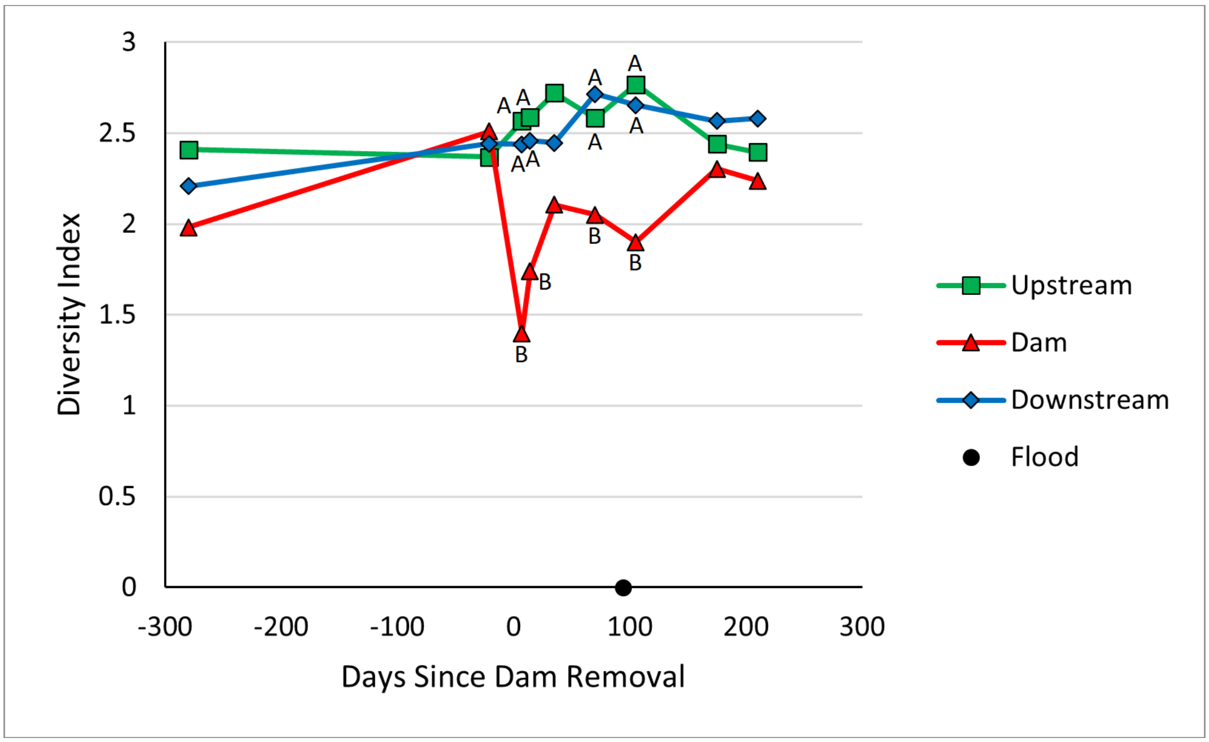

Figure 28: *Shannon diversity at the Family taxonomic rank for the three groups. One week post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two weeks post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

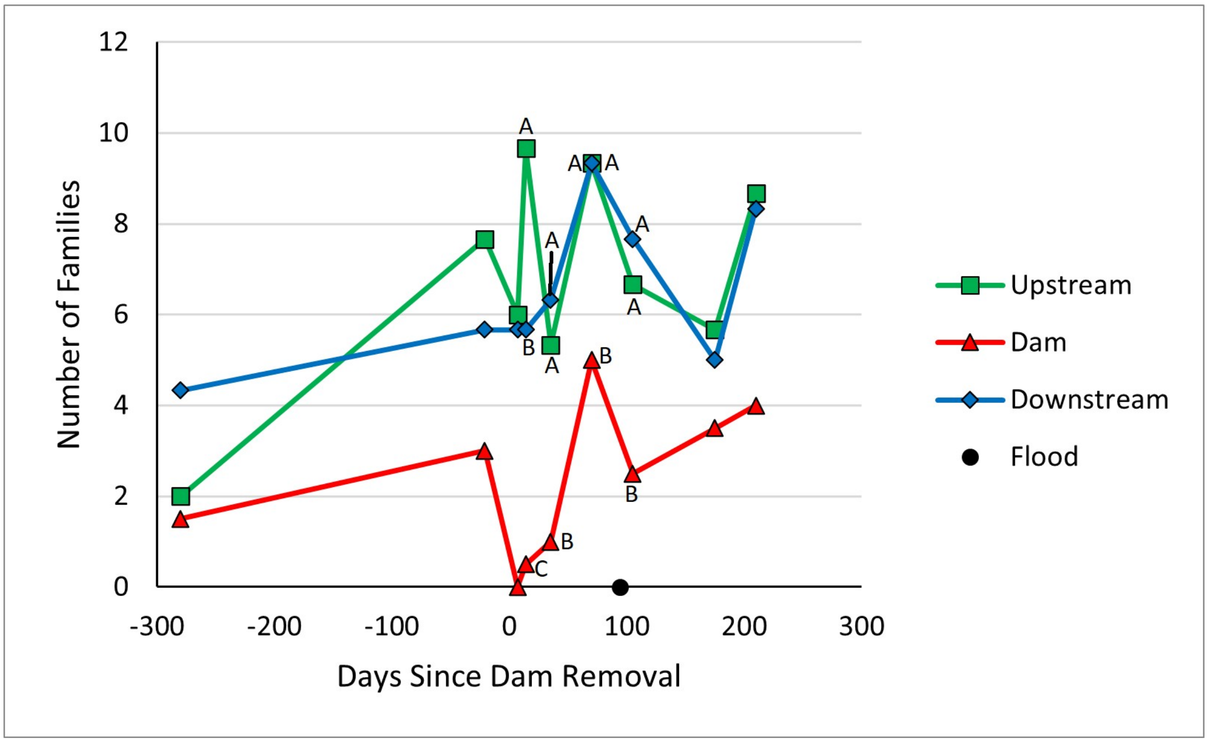

Figure 29: *Trichoptera richness at the Family taxonomic rank for the three groups. At the fall pre-removal survey the Downstream group differs from both the Dam and Upstream, but they do not differ from each other. Two weeks post removal all three groups differ from each other. One month post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

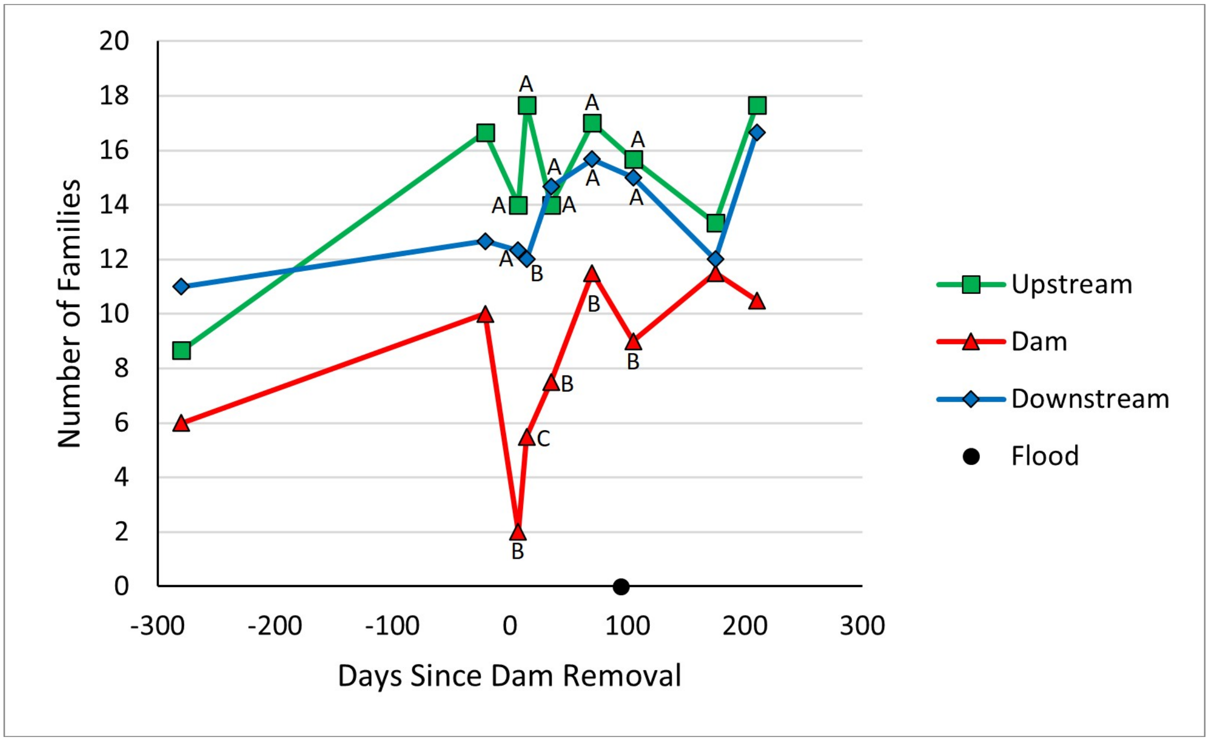

Figure 30: *Combined Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness at the Family taxonomic rank for the three groups. One week post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two weeks post removal all three groups differ from each other. One month post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

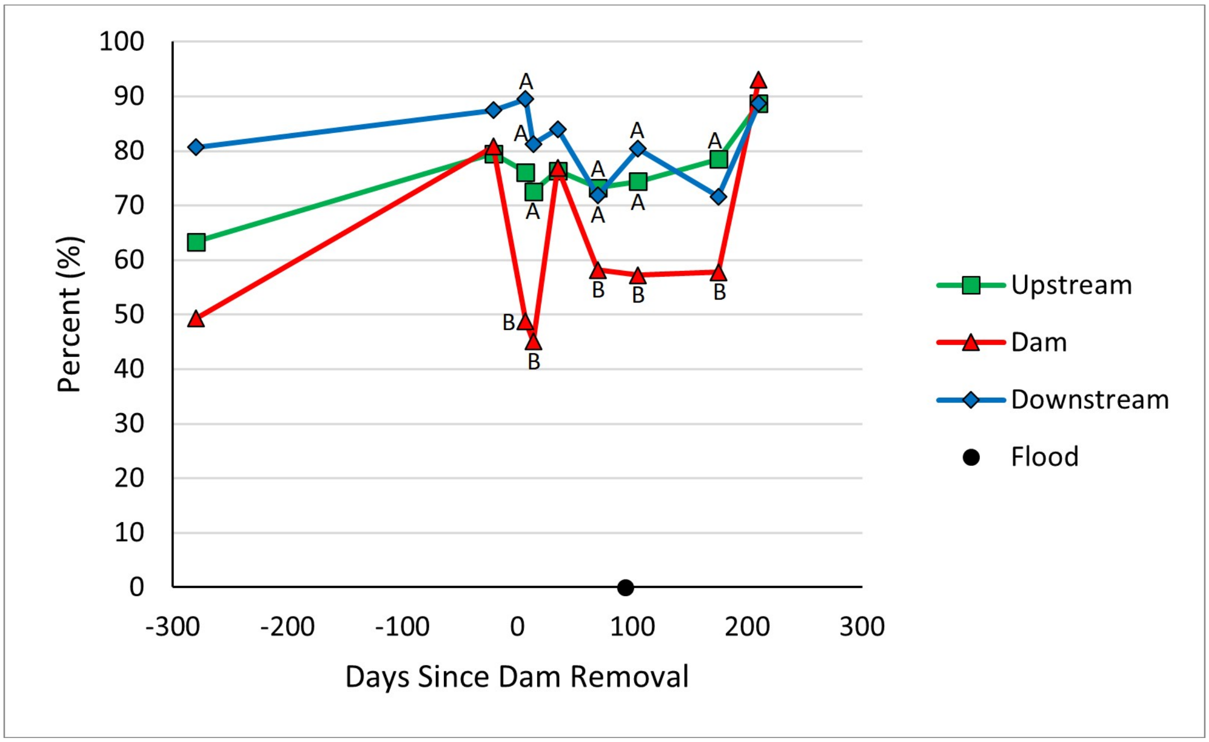

Figure 31: *Percentage of benthic macroinvertebrates with the Clinger functional habit for the three groups. One week post removal the Dam site differs from Downstream, but neither differs from Upstream. Two weeks post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Four months post removal the Dam site differs from Upstream, but neither differs from Downstream.*

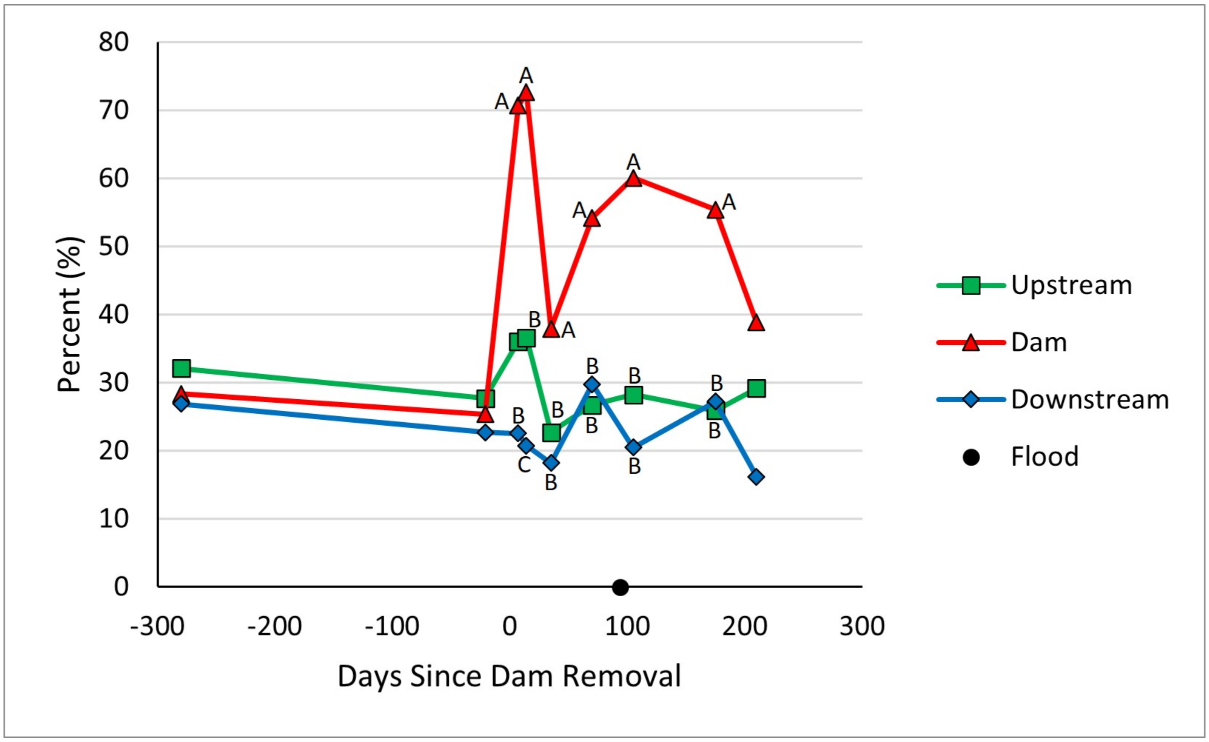

Figure 32: *Percentage of benthic macroinvertebrates with a multivoltine life history for the three groups. One week post removal the Dam site differs from Downstream, but neither differs from Upstream. Two weeks post removal all three groups differ from each other. One month post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Four months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

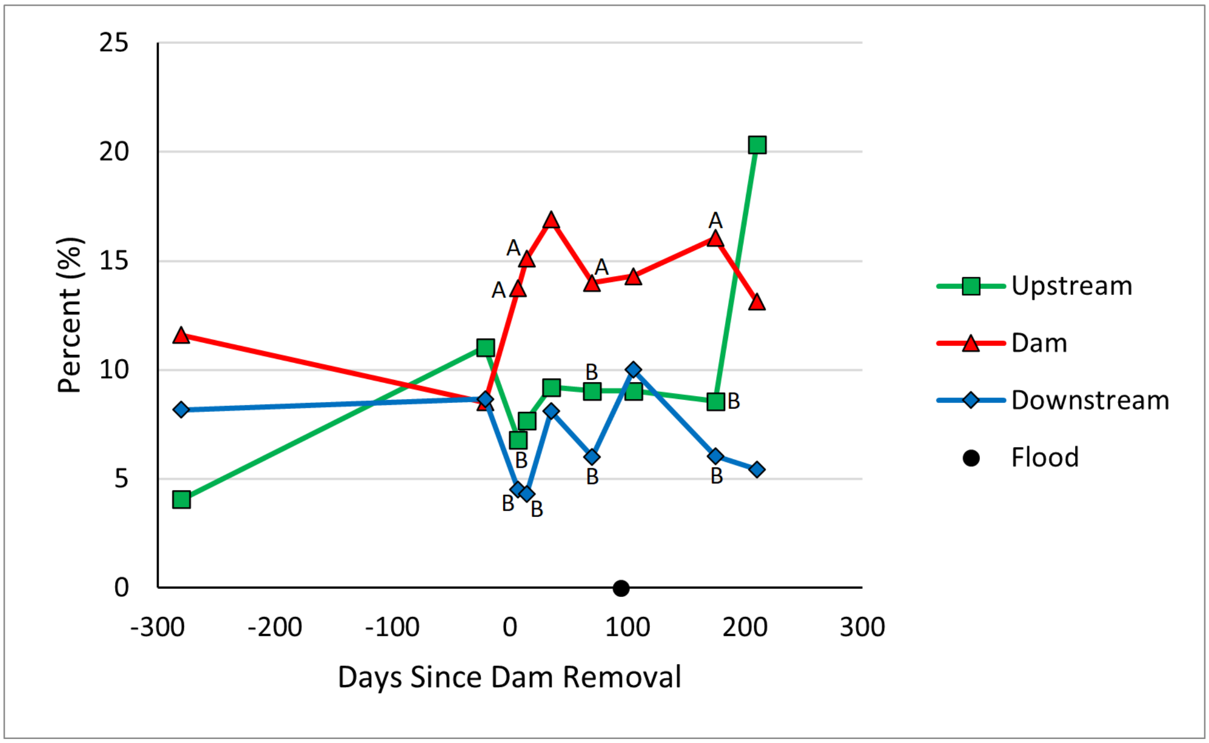

Figure 33: *Percentage of Diptera among collected benthic macroinvertebrates for the three groups. One week post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two weeks post removal the Dam site differs from Downstream, but neither differs from Upstream. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Four months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

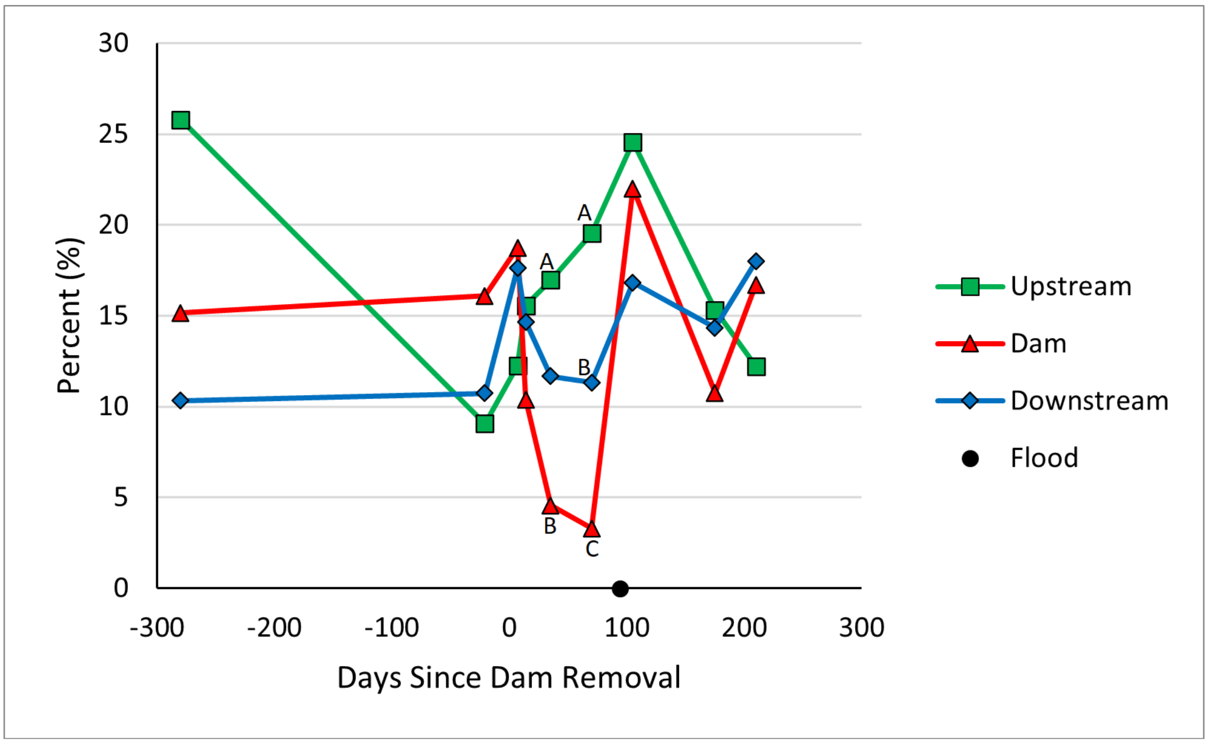

Figure 34: *Percentage of benthic macroinvertebrates with the Scraper functional feeding style for the three groups. One month post removal the Dam site differs from Upstream, but neither differs from Downstream. Two months post removal all three groups differ from each other.*

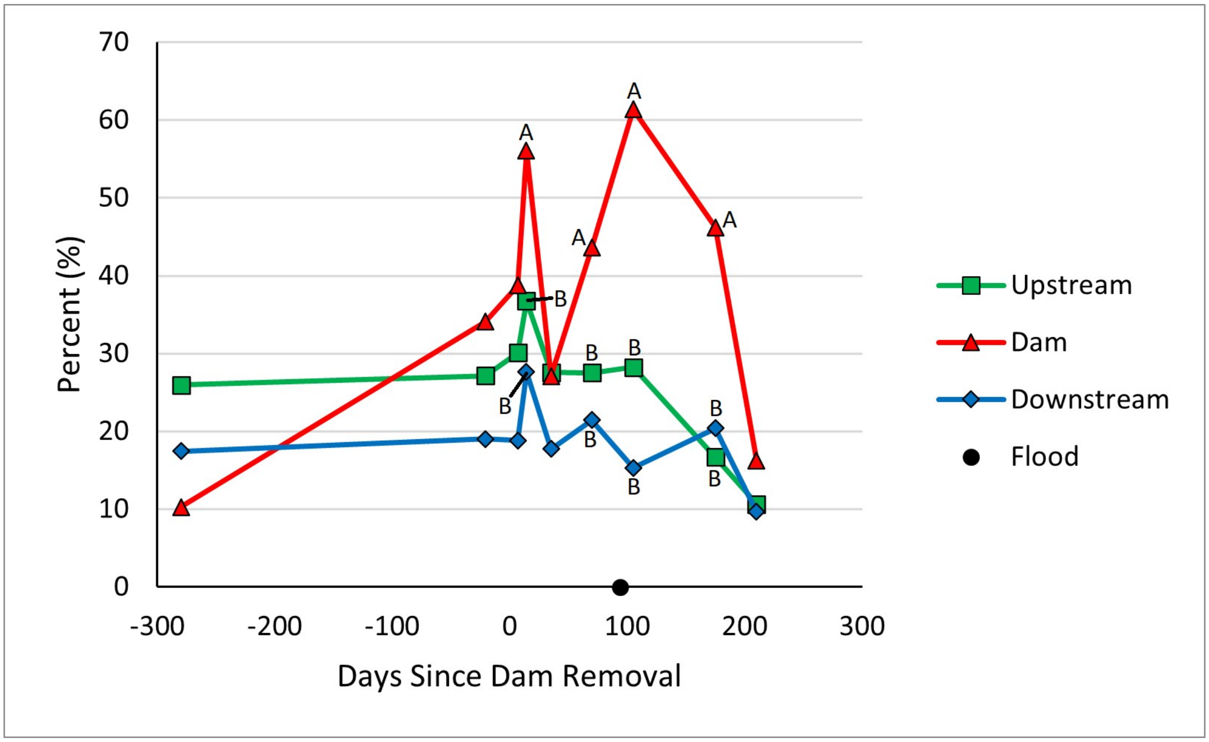

Figure 35: *Percentage of Ephemeroptera among collected benthic macroinvertebrates for the three groups. Two weeks post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Two months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Three months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other. Four months post removal the Dam site differs from both Upstream and Downstream, but they do not differ from each other.*

Figure 36: *Statistically and ecologically significant metrics at the Downstream Impacted sites as compared to Upstream Reference sites. Black horizontal bars indicate the duration of significance. Trichoptera richness and EPT richness were significantly lower at 2 Weeks, and not significantly different at 1 Month. Unexplained D_{50} increase at the Hubert Thomas Road bridge at the 4 Month survey resulted in loss of statistical significance and data dropout at the US Hwy. 321 bridge at 6 Months prevented analysis. Return to low D_{50} at Hubert Thomas and maintained low D_{50} at the Pasture Site, as well as recent field observations indicate continued decrease of D_{50} for the Downstream Impacted group despite no statistical difference relative to Upstream Reference sites.*

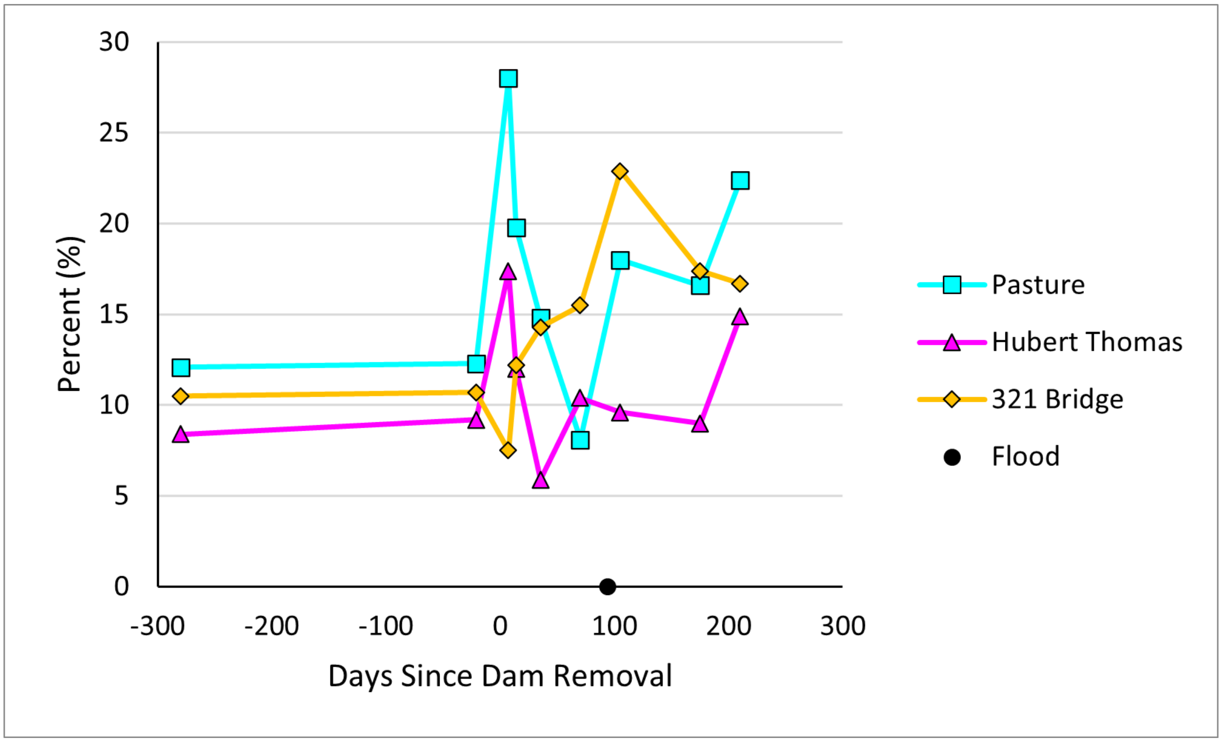

Figure 37: *Percentage of benthic macroinvertebrates with the Scraper functional feeding style at the three Downstream Impacted sites.*

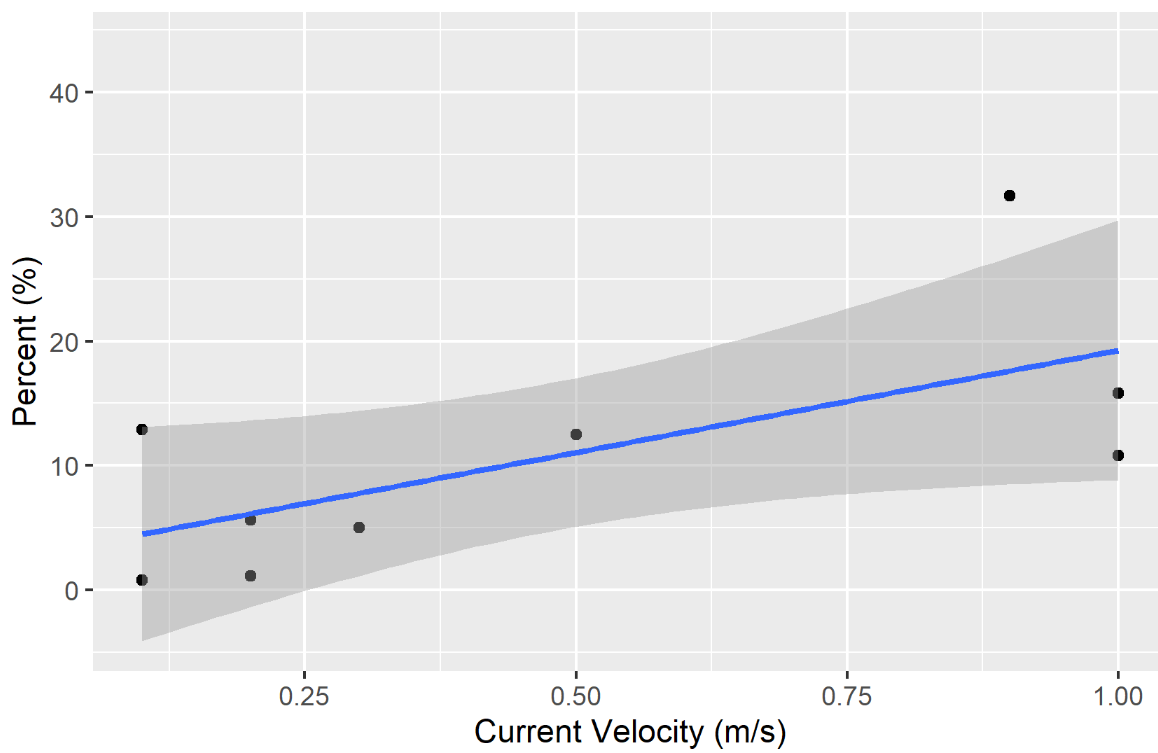

Figure 38: *Percent Plecoptera as a response variable to Bed Velocity.* $R^2 = 0.445$, $p = 0.050$, $y = 16.4x + 2.9$

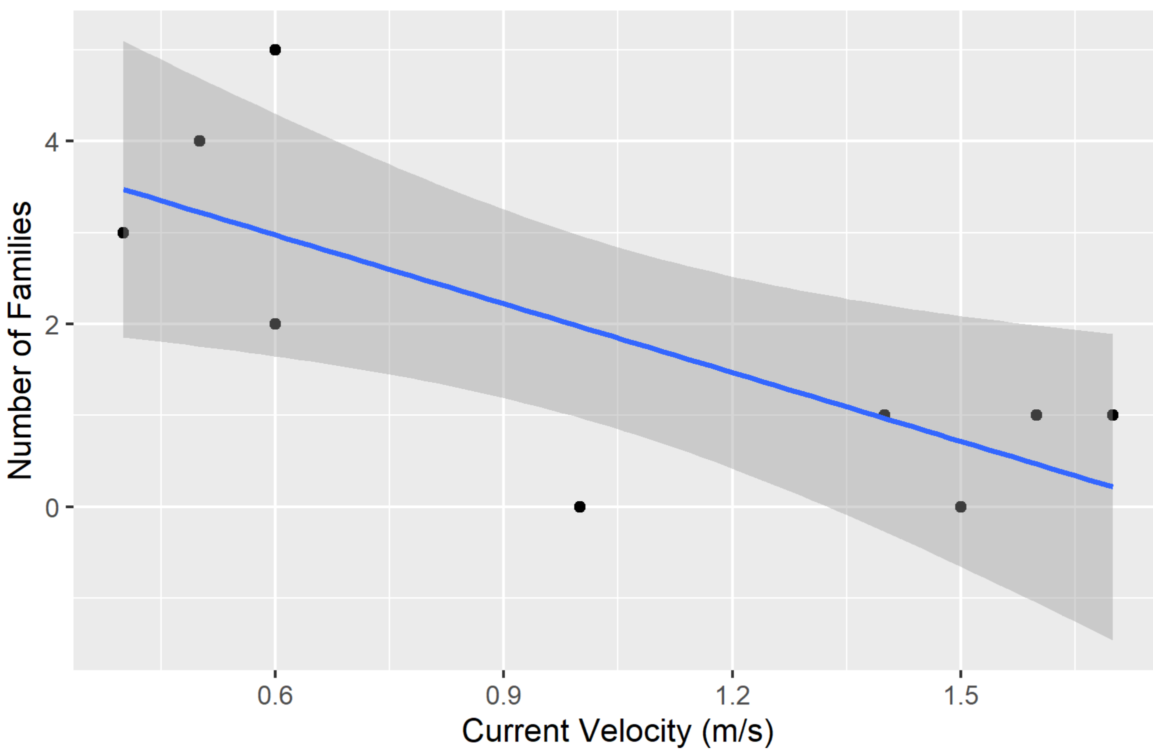

Figure 39: *Trichoptera Richness as a response variable to 40% Depth Velocity.* $R^2 = 0.551$, $p = 0.022$, $y = -2.5x + 4.48$

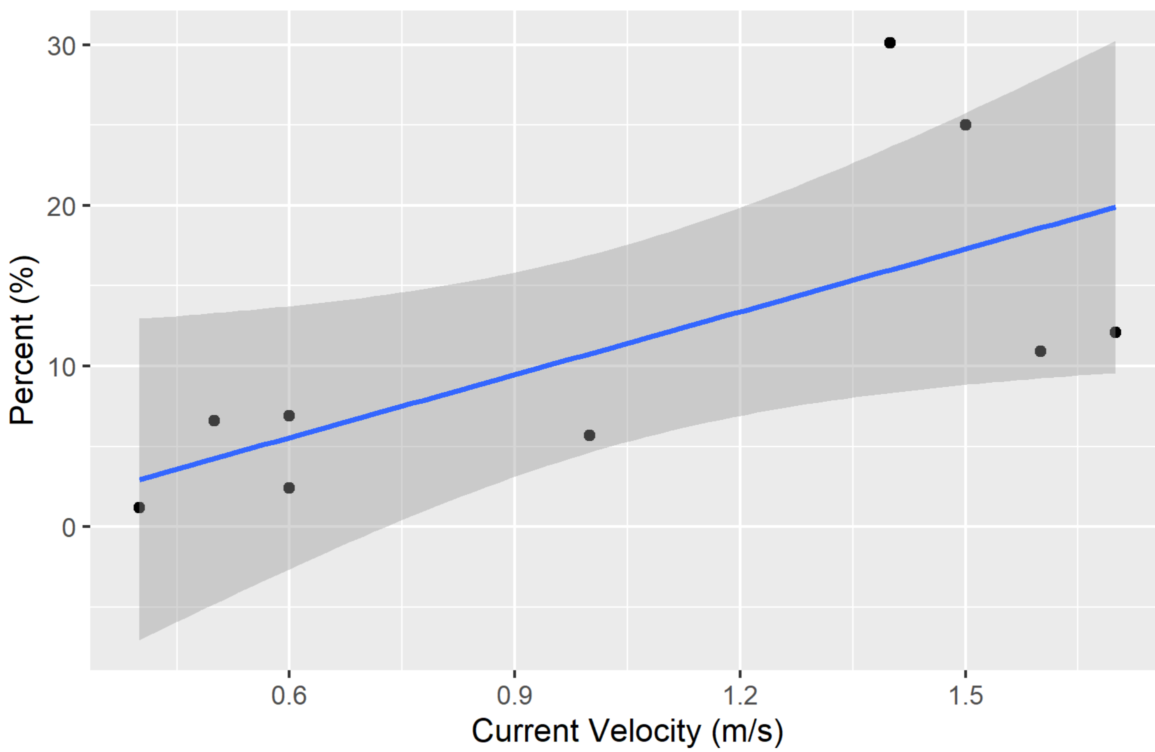

Figure 40: *Percent Semivoltine as a response variable to 40% Depth Velocity.* $R^2 = 0.466$, $p = 0.043$, $y = 13.1x - 2.3$

Vita

William Gregg McMahan was born in Salem, Virginia to Ms. Paula and Mr. Gregory McMahan. He graduated from Fayette County High School in Fayetteville, Georgia, in May of 2012. In August of 2012 he enrolled in the Corps of Cadets at North Georgia College and State University as a Dual Degree Physics/Mechanical Engineering Major. He transferred to Appalachian State University in January of 2016 and earned a Bachelor of Science in Environmental Geology in December of 2018. In January of 2019 he began Graduate School at Appalicahin State University under the tutelage of Dr. Shea Tuberty, and earned a Master of Science in Ecology in May of 2023.

In his free time William is a member of the Watauga County Rescue Squad, the Linville Central Rescue Squad, and the North Carolina Mountain Rescue Team. He enjoys fly fishing, whitewater kayaking, rock climbing, and searching for hidden waterfalls far from established trails.