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## **Quantifying the success and long-term ecological and socioeconomic benefits of watershed-scale acid mine drainage (AMD) remediation efforts within West Virginia**

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# Abstract

Stream restoration projects are increasingly common. However, restoration projects that establish measurable goals, have pre- and post-restoration monitoring, and are implemented at the watershed scale are rare. We conducted a long-term (9-year) before-after-control-impact designed assessment of two watershed-scale acid mine drainage (AMD) remediation programs, one in a warm-water ecosystem and one in a cold-water ecosystem in West Virginia, USA. The restoration was strategically designed to recover biodiversity and improve the native fisheries by restoring chemically degraded water quality and re-establishing riverscape connectivity. We used repeated-measures analysis of variance to quantify responses in water chemistry, benthic macroinvertebrate communities, and fish community composition before and after restoration within and among treated and untreated sites. Warm- and cold-water watersheds exhibited significant improvements post-restoration in water quality and macroinvertebrate communities in both watersheds. However, differences in fish community responses indicate that regionally degraded conditions may play a role in the ability of fish communities to recover in restored systems. Fish diversity increased to reference conditions in both watersheds, but functional fisheries are not recovering. In the warm-water system, the reference sites do not meet the regional drainage area to species richness relationship whereas the cold-water system has intact reference populations within the watershed. This suggests that successful fishery restoration in degraded watersheds depends on the presence of a regional species pool available to repopulate the targeted watershed. Furthermore, long-term changes in fish communities in the cold-water system indicate that fish populations may have a delayed response to restoration projects. Treated sites within the cold-water watershed had significant improvements in water chemistry and macroinvertebrates from 2008 (i.e., pre-restoration) to 2013 and remained unchanged from 2013 to 2017. However, fish diversity and brook trout populations in treated streams increased significantly from 2008 to 2013 and continued to increase in both 2017 and 2018. The continued increase in brook trout populations over time suggests that restoration was successful in reestablishing connectivity among restored and previously intact brook trout sub-populations. Consequently, the full benefit of restoration may not yet be realized as fish populations continue to expand.

In this study, we 1) characterized the long-term ecological response to two state-of-the-art watershed-scale AMD treatment efforts; 2) quantified temporal changes in success of each restoration effort; and 3) developed a remediation prioritization framework for AMD-impacted systems that simultaneously maximizes ecological and socioeconomic benefits.

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# Executive Summary

Appalachia has suffered from many degraded streams due to historical mining conditions that were left neglected. Treatment activities that only consider the local reach-scale do not fully capture the watershed-scale benefits that can accrue to biological systems. Restoration and reclamation efforts should begin with a focus on watershed-scale benefits.

When considering efforts to restore acid mine drainage (AMD)-impacted systems, there remains considerable uncertainty as to the long-term ecological and societal benefits--particularly for projects occurring across larger spatial (i.e., watershed) scales. This represents a critical knowledge gap because successful restoration requires an adaptive management framework wherein remediation activities are first prioritized and later altered based on projected and observed ecological response and socioeconomic outcomes.

The *specific objectives* of this project were to: 1) characterize long-term ecological response to two state-of-the-art watershed-scale AMD treatment efforts; 2) quantify temporal changes in success of each restoration effort; and 3) develop a remediation prioritization framework for AMDimpacted systems that simultaneously maximizes ecological and socioeconomic benefits.

During this project, ongoing efforts to develop, implement, and assess two watershed-scale AMD remediation plans (Abrams Creek and Three Fork Creek watersheds) were extended. Specifically, we extended a study that utilizes a watershed-scale, before-after-control-impact sampling design to empirically quantify and statistically test for ecological response to remediation efforts over time. We integrated existing data and newly collected data into an unprecedented 10 year dataset on watershed-scale AMD remediation efforts. Using this dataset, we characterized and quantified long-term watershed-scale changes in physicochemical conditions and aquatic community structure in response to AMD remediation. We then utilized a unique method of quantifying the ecological value of stream segments, called Ecological Units (EUs), to obtain an objective, scientifically-based measure of long-term remediation success in each watershed. Finally, we integrated EUs with newly collected data on recreational use of remediated systems into a prioritization framework that facilitates management decisions within AMD-impacted watersheds by simultaneously maximizing ecological and socioeconomic benefits. This framework is transferrable to other regions throughout the United States impacted by any number of anthropogenic disturbances.

In general, our research results can be used by OSMRE and other regulatory agencies to address water quality impacts by:

- Providing a better understanding of the long-term effects that mining and reclamation practices have on physical, chemical, and biological aquatic resources;
- Providing an unprecedented dataset that will allow the first long-term assessment of current remediation efforts that utilize watershed-scale strategies and active treatment technologies; and
- Providing a framework through which resource managers can both prioritize remediation activities and determine the success of implemented remedial measures based on both ecological and socioeconomic benefits and perspectives.

Specifically, our results indicate the following:

- Restoration projects focused at the watershed scale are expected to be more successful than improving local conditions alone.
- Based on results at Abrams Creek, it may take longer timeframes to see the full benefits of restoration and for biological communities to fully recover.
- It is important to consider regional context in order to understand systems; this is due to potential impacts that suppress fish communities.
- The recovery of Three Fork Creek may not be feasible due to regional impairments.
- These results can help watershed managers recognize the importance of long-term monitoring and regional-scale planning to improve the design and implementation of restoration projects.

## Introduction

Legacy effects of mining on aquatic ecosystems are among the most critical environmental issues currently facing the US. However, there remains considerable uncertainty as to the long-term ecological and societal benefits of efforts to restore mining-impacted systems, particularly projects occurring across larger spatial scales. This represents a critical knowledge gap because successful remediation of AMD-impacted watersheds will require adaptive management systems wherein remediation activities are altered based on observed ecological and socioeconomic responses to remediation activities over time. The research seeks to quantify long-term, watershed-scale ecological and socioeconomic benefits of state-of-the-art AMD remediation efforts within two central Appalachian watersheds – Abrams Creek and Three Fork Creek. This research is part of an integrated effort to develop the scientific and technical tools needed to manage mining-related impacts and ensure maintenance, restoration, and long-term protection of aquatic ecosystem services in AMD-impacted Appalachian watersheds.

Adaptive management was used in this study as our structured decisionmaking framework to address water quality issues in the study area watersheds. Because management decisions require a high level of certainty with data and models, it was imperative

capture the best



*Figure 1. Illustration of the adaptive management framework designed for remediation of AMD watersheds. Red numbers indicate objectives.*

available data that was both spatially and temporally relevant. The adaptive management framework (Figure 1) allowed us to account for data to match the goals and objectives that can result in sustainable policies for the restoration and mitigation of impacted systems. Throughout this project, the adaptive management framework helped to structure management objectives to guide decisions about what actions to take and explicit assumptions about expected outcomes to compare against actual outcomes.

Historically, Abrams Creek and Three Fork Creek (Figures 2-4) have sustained productive coldand warm-water fisheries until coal mining entered the region in the early 1900s (NFWP/FOB 2009). In Abrams and Three Fork, AMD is often generated by abandoned mine lands (AMLs): coal mines that were abandoned prior to the Surface Mining Control and Reclamation Act (SMCRA) of 1977. Mines in operation since that time are more closely regulated and cannot legally discharge waste of the same toxicity. Mining companies must pay a bond up front before beginning operations, which is only refunded upon satisfactory remediation of the mining site. However, some mining companies choose to forfeit their performance bonds instead of meeting their SMCRA permit requirements, creating bond forfeiture sites (BFS). This often happens when the cost of remediation is greater than the value of the bond. Both AMLs and BFS can discharge AMD, and it is generally the responsibility of government agencies, often with the support of local watershed organizations and other partners, to remediate these sites (NFWP/FOB 2009).

Biological stream health is often measured by indices based on the diversity and size of benthic macroinvertebrate or fish populations. If the stream beds become an inhospitable habitat for reproduction, and if the stream water itself has high concentrations of harmful chemicals, then the aquatic life in the streams will be harmed. It is not uncommon to find only acid‐tolerant insect species or to find stream reaches with no fish downstream from AMD sources. AMD can also harm native plants, dissolve bridge supports and pipes, smell foul, and detract from the natural aesthetic value of impacted waterways.

Consequently, fishery restoration was the primary objective of each of the associated AMD remediation projects. This is notable since restoration practitioners have generally undersold the evidence of benefits of restoration as a worthwhile investment for society (Arson et al. 2010).

Natural processes that occur in small streams and wetlands provide humans with a host of benefits, including flood control, maintenance of water quantity and quality, and habitat for a variety of plants and animals. For headwater streams and wetlands to provide ecosystem services that sustain the health of our nation's waters, the hydrological, geological and biological components of stream networks must be intact (Meyers et al. 2013). We wanted to highlight the importance of headwater catchments by focusing on the quantity and value of ecosystem services derived from them and to extrapolate that importance from regional to national scales within the continental United States. We focused on headwaters because that is a particular category of streams that is of interest to the US regulatory community. As an under-protected resource, we wanted to highlight their value. We combined data collected from headwater streams as a part of the US Environmental Protection Agency's (USEPA) National Rivers and Streams Assessment (NRSA) with catchment attributes related to the water supply. We used these data to develop ecological production functions related to the delivery of ecosystem services from headwater catchments and combine these services with published valuations to estimate potential cumulative benefits derived from headwater catchments in the United States. We captured these measures using the benefit transfer approach once we determined that onsite surveys would not produce adequate results due to the rural and remote nature of these sites.

The specific objectives of the project are to:

- Quantify long-term ecological response to two watershed-scale AMD treatment efforts (Abrams Creek and Three Fork Creek) that integrate multiple treatment activities and technologies (i.e., active and passive).
- Quantify long-term success of watershed-scale restoration efforts within Abrams Creek and Three Fork Creek by comparing observed and predicted response.
- Develop a remediation prioritization framework for AMD-impacted systems that simultaneously maximize ecological and socioeconomic benefits.

## **Experimental**

Objective 1 – Quantify long-term ecological response to two watershed-scale AMD treatment efforts (Abrams Creek and Three Fork Creek) that integrate multiple treatment activities and technologies (i.e. active and passive)

### Site selection

This project followed a beforeafter-control-impact (BACI) sampling design. Sites were classified into three categories: untreated streams impaired by AMD (AMD sites), streams treated for AMD (treated sites), and unimpaired reference streams (reference sites).



*Figure 2. Map of Three Fork Creek, West Virginia with site locations, site types (i.e. treated, AMD, and reference), and treatment locations.*

Targeted sites were strategically chosen based on treatment locations and stream confluences. In Three Fork Creek, 17 sites were sampled in 2008 prior to treatment for AMD. An additional three unimpacted reference sites were added before late-summer fish sampling in 2008. We sampled the same sites with the additional three unimpacted reference sites in 2017 and in 2018 for all data collections post-restoration (Figure 2; Table 1).



*Table 1. Information for sites in the Three Fork Watershed: Site names, GPS coordinates, drainage area in km2, and stream type: AMD (unremediated), Treated (remediated), or Reference (unimpaired).*

*Table 2. Information for the sites in the Abrams Creek watershed: Site names, GPS coordinates, drainage area in km2, and stream type: AMD (unremediated), Treated (remediated), or Reference (unimpaired).*



Fourteen sites were sampled in Abrams Creek in spring 2008 for water chemistry and macroinvertebrate sampling. An additional three unimpacted reference stream sites and one AMD site were added before fish sampling in the late-summer of 2008. The same sites, plus the additional four sites, were sampled again in 2013 postrestoration for all data collections. All 18 sites were sampled again in 2017 and 2018 (Figure 3; Table 2).

## Ecological Data **Collection**

We integrated pre-existing data with newly collected data into a long-term temporal dataset.

### Pre-existing data

Pre-existing data came from several sources: 1) watershed assessments conducted by the WVDEP (both the AML and



*Figure 3. Map of Abrams Creek, West Virginia with site locations, site types (i.e. treated, AMD, and reference), and treatment locations.*

Watershed Assessment Branches); 2) our own sampling efforts over the past seven years (Petty et al. 2008; Strager et al. 2008; WVWRI 2007; Watson et al. 2018).

### New data

Reach lengths were defined as 40 times the mean stream width, with minimum and maximum lengths of 150m and 300m. We collected a comprehensive suite of physical, chemical, and biological data at all previously sampled locations (Figure 4). Overall physical habitat quality and complexity were evaluated using USEPA Rapid Visual Habitat Assessment (RVHA) protocols (Barbour et al. 1999). We measured reach-scale complexity by taking measurements of water depth, channel-unit type (riffle, run, pool, glide), and distance to nearest fish cover (defined as any structure within the active channel capable of concealing a 20.32-cm fish) at evenly spaced points along the thalweg (Petty et al. 2001). Large woody debris was counted and categorized based on diameter and length (Petty et al. 2001). We conducted a modified Wolman pebble count to categorize 100 randomly chosen substrate particles (Wolman 1954; Merriam et al. 2011).



*Figure 4. Location of Abrams Creek and Three Fork Creek within West Virginia. Also shown are implemented remediation actions that include active in-stream dosing, passive treatment systems, and limestone sand additions. Treatment plans were developed by investigators and implemented by WVDEP. Locations of pre- and post-treatment assessment sites are shown for each watershed.*

### Benthic macroinvertebrates

Benthic macroinvertebrates were collected following the WVDEP's standard operating procedures. Samples were collected via kick net collection at four riffles, which yields one square meter of total sampled area for each site (WVDEP 2018). Contents were combined and immediately preserved with 95% ethanol. Samples were later subsampled to 200 individuals (pursuant to WVDEP protocol) and identified to genus using the 4th edition of Merritt and Cummins' dichotomous key, "An Introduction to the Aquatic Insects of North America." To quantify macroinvertebrate response to restoration, we calculated the Genus-Level Index of Most Probable Stream Status (GLIMPSS: Pond et al. 2008; Pond et al. 2013), which is a genus-level macroinvertebrate index of biotic integrity for wadeable streams in West Virginia.

### Fish

Fish assemblages were sampled at each site following WVDEP standard procedures during late summer baseflows (WVDEPa 2013). Fish sampling was completed in both watersheds between mid-July and mid-September in 2008, 2013 (Abrams Creek only), 2017, and 2018 using the same methods for both watersheds. We used one-pass backpack electrofishing techniques for all sites.

One to three backpacks were used depending on stream size. Sample reach lengths were 40 times the mean stream width with a minimum of 150m and a maximum of 300m. All individuals were identified to species and released.

### Water chemistry

Water chemistry was sampled during spring baseflows in 2008, 2013 (Abrams Creek only), 2017 and 2018. Samples were collected between mid-May to mid-June each sampling year for both watersheds. All samples were collected using the same methods across both watersheds for each sampling year. We collected insitu measurements of pH, conductivity, temperature, and dissolved oxygen at each site using a YSI 600 XLM multiparameter probe at each sampling location. We collected grab samples at the same sample sites and stored them at 4°C until the analysis was completed at the National Research Center for Coal and Energy Laboratory at West Virginia University. The samples were analyzed for alkalinity/acidity, sulfate, in addition to total and dissolved aluminum, barium, copper, chloride ion, cobalt, chromium, cadmium, calcium, sodium, nickel, selenium, zinc, iron, magnesium, and manganese concentrations (mg/L).

## Objective 2 – Quantify the long-term success of watershed-scale restoration efforts within Abrams Creek and Three Fork Creek by comparing observed and predicted response.

## Restoration Success

During the planning phase of these restoration projects, methods that give the watershed an "ecological currency" were used to determine the best and most economical restoration project plan (Petty et al. 2008). This method uses a measurement tool called "EcoUnits" (EUs) to quantify useable stream miles for specific functions (Petty and Thorne 2005; Merovich and Petty 2007; Petty et al. 2008; Poplar-Jeffers et al. 2009; Watson et al. 2017). Stream segment lengths are weighted by ecological function ranging from zero to one. A high-quality stream segment with a weighting of one indicates it is reaching 100% of what is expected of high-quality streams in the region. Stream segments with ratings of zero indicate the stream is highly impaired and not functioning ecologically.

In Three Fork Creek, we calculated four EUs: diversity EU, cold-water fishery EU, warm-water fishery EU, and overall fishery EU (Petty et al. 2008). In Abrams Creek, EUs were calculated for macroinvertebrate diversity, brook trout fishery, stocked trout fishery, and overall fishery (Watson et al. 2017). A quantitative, repeatable, and robust measure of biological conditions known as an index of biotic integrity (IBI), the West Virginia Stream Condition Index (WVSCI), was used to determine ecological conditions for each segment-level watershed for each measured function with condition weightings found in Petty et al. 2008 (Three Fork Creek) and Watson et al. 2017 (Abrams Creek). EUs were calculated for each segment level watershed using these condition scores against their ecological potential for each segment. With this method, we obtained historical, predicted, pre-restoration, and current EUs for each segment, which could be combined into cumulative EUs for each watershed. Kolmogorov-Smirnov (KS) tests were used to compare current EUs to predicted post-restoration EUs within each watershed.

## Objective 3 – Develop a remediation prioritization framework for AMDimpacted systems that simultaneously maximizes ecological and socioeconomic benefits.

### Quantify and characterize ecosystem services

### Background Research

Ecosystem services are benefits that humans acure form the environment and natural resources that are not easily identified because they are not bought and sold during market transactions. Environmental economics was developed to better understand and account for the many nonmarket benefits that can be occur from ecosystem services. One method to estimate benefits is called benefit transfer. The benefit transfer approach involves the spatial and temporal transfer of economic information captured from one site to make inferences about the economic value of environmental goods and services of another site. In this approach, the location where the original study is conducted is called a 'study site,' whereas a 'policy site' is considered for benefit transfer and is usually part of an economic analysis of proposed policy action (Bergstrom and DeCivita 1999). Due to a lack of time and resources or high costs of conducting primary observational research, the benefit transfer approach has become popular among researchers studying the recreational uses of natural sites (Rosenberger and Loomis 2001). Decision-makers have found timely and low-cost methods to assign monetary values to the benefits of goods and services received from ecosystem services.

Due to evolution in approaches, the calculations of benefit transfer have changed. Earlier works distinguished this approach under three broad categories: Unit or fixed value transfer, (2) transfers adjusted using expert judgments, and (3) function transfer (Brookshire and Neill 1992; Desvousges et al. 1992; Bergstrom and DeCivita 1999). Based on this approach, economic estimates are either transferred as monetary value units or as value functions conditioned on explanatory variables that define the attributes of an ecological and economical choice setting.

Chronologically recent works treat transfer method, earlier a part of expert judgments, unmistakably distinct from unit value. Now, four different benefit transfer methodologies exist in the literature: Benefit estimate transfer, benefit function transfer, meta-analysis function -- and the most recent one – preference calibration transfer (Smith et al. 2006). Using these methodologies, the benefit for policy sites are obtained from study sites based on stated and revealed preference estimation methods (Barton 2002). For example, Scarpra et al. (2000) and Matthews et al. (2009) conduct unit value transfer tests. The objective of their studies is to investigate the reliability of estimates obtained from transferring benefits based on the contingent valuation of forest recreation conditional on forest-specific attributes. Studies testing the accuracy of benefit transfer function across study and policy studies include Barton (2002). Adding to this pool of literature,

Groothuis et al. (2005) analyze estimates of travel cost and contingent valuation to gain insights into the benefit transfer approach. In this study, Groothuis et al. (2005) perform a comparison between the unit value and benefit transfer function for the transferability of economic benefits between two sites (study and policy) in wildlife recreational setting. Kaul et al. 2013 use a nonparametric approach to meta-analysis to identify modeling decisions affecting benefit transfer error.

For this study, we explored reasons why benefit transfer is useful and popular among researchers. On the downside, several factors affect the reliability and validity of the benefit transfer approach. Quality of original study significantly impacts studies carried out on policy site, thereby affecting outcomes of policy site. This can also be termed as garbage-in, garbage out factor. On other occasions, information is drawn from a limited pool of sources, typically investigating the economic value of study sites.

Critics believe that benefit transfer lacks a micro-level theoretical foundation as circumstances at the study site, and policy site might not remain the same throughout. Moreover, estimates of demand functions derived from changes in environmental quality are dependent on specific site attributes, preferences, and demographics of the site. All these factors are likely to change over time (Kirchhoff et al. 1997). Even if these characteristics remain the same, the problem of inconsistency takes place.

### Approach

Our approach to calculating ecosystem services has led us to the use of the Benefit Transfer. Applying some monetary estimates from previous studies, we found a study conducted by Mazzotta et al. (2015) in a HUC-10 watershed area in West Virginia. This area was affected by surface mining and was predicted to have a loss of 0.87% of gamefish abundance in a partial mining scenario and almost five times the loss in a full mining scenario. Stauffer and Ferreri (2002) and Hopkins and Roush (2013) also report the loss of fish species attributed to mining. The total annual welfare losses were calculated as \$120,500 for the partial scenario and \$627,800 for the full scenario due to changes in recreational fishing catches (Mazzotta et al. 2015). Bergstrom and Cordell (1991) report the total value of outdoor activities modeled ranged from \$267 million to \$16 million annually. Few studies have also evaluated per household WTP to protect the type of species. Loomis and White (1996) explain the uses of WTP values to show that over half of the variation in WTP is explained by the change in the size of the population, payment type, frequency of visits, and species type. Their results show that annual WTP ranges from a low of \$6 per household for fish such as the striped shiner to a high of \$95 per household for the northern spotted owl and its old-growth habitat. Other than fishing, swimming is also one activity that is valued highly on the recreational activity list. The application of meta-analysis function provides a national average measure of the benefit of swimming value of \$14.44 per person per day on average across the US (Rosenberger and Loomis 2001).

To calculate the ecosystem services, we built and used a database of variables and benefit values from peer-reviewed literature. For each variable used in the regression model, we made an original goal to find at least three peer-reviewed papers, preferably from different regions, to represent a cross-section of study and research in the areas. However, due to the benefit transfer technique being a relatively new resource economic evaluation technique, we were only able to find at most two papers for each variable partially attributed to North America watersheds and ecoregion systems. We do not necessarily see this as a limitation of our work but an opportunity to expand this field of study and identify a research need for future work. We constructed a meta‐ regression model to identify a benefit transfer valuation that relates the ecosystem service of a remediated water body to its physical, demographic, economic, and geographic characteristics. A meta-regression model integrates findings from multiple primary studies of a common amenity. An attractive feature of meta-analysis is the ability to control for features that are fixed for any given study but vary across studies (Nelson and Kennedy 2009). This data richness is what provides a meta-function with the best opportunity to calibrate value predictions to policy site conditions.

# Results and Discussion

## Objective 1 – Quantify long-term ecological response to two watershed-scale AMD treatment efforts.

### Ecological Response

We used multivariate tests and ordination procedures to characterize and quantify long-term ecological responses to remediation activities. In this task, we used a series of univariate and multivariate statistical techniques to characterize and quantify long-term watershed changes in physio-chemical conditions and aquatic community structure as a function of AMD remediation.

### Benthic macroinvertebrates

To quantify macroinvertebrate response to restoration, we calculated the Genus-Level Index of Most Probable Stream Status (GLIMPSS; Pond et al. 2013), which is a genus-level macroinvertebrate index of biotic integrity for wadeable streams in West Virginia.

We used analysis of variance (ANOVA) to test for significant increases in GLIMPSS scores of treated sites since restoration in Three Fork Creek (Table 3) followed by Tukey post-tests to compare index scores of treated and reference streams post-restoration. We used two-way ANOVA to test increases in GLIMPSS scores within treated sites post-restoration in Abrams Creek, followed by Tukey post-tests to compare treated and reference scores (Table 4). WVSCI and GLIMPSS were averaged for each watershed by treatment type; a higher value is better within both indices (Table 5).



*Table 3. GLIMPSS scores for each site sampled in Three Fork Creek for the 2008 (pre-restoration) and 2017-18 (post-restoration) sampling years.*

*Table 4. GLIMPSS scores for each site sampled in Abrams Creek for the 2008 (pre-restoration), 2013, 2017 and 2018 (post-restoration) sampling years.*

<b>Abrams</b>		<b>GLIMPSS</b>			
<b>Site</b>	<b>Type</b>	2008	2013	2017	2018
Emory Creek Headwater Left Fork	AMD		19.33	62.82	23.52
Unnamed Tributary 1 Emory Creek	AMD			56.09	61.92
<b>Emory Creek Headwater Right Fork</b>	reference		61.38	74.35	77.72
Johnnycake at Mouth	reference	77.15	76.19	68.3	91.55
Unnamed Tributary 2 mory Creek	reference		69.19	58	56.29
Upper Johnnycake Run	reference		84.4	82.7	83.87
Abrams Creek above Emory	treated	61.14	63.8	68.52	71.95
Abrams Creek above Glade	treated	19	39.45	43.39	56.23
Abrams Creek above Johnnyckae	treated	20.67	58.02	47.55	44.41
Abrams Creek above Laurel	treated	16.13	58.66	56.22	44.57
Abrams Creek at CR 42	treated	3.73	32.51	49.89	41.1
<b>Abrams Creek at Laytons</b>	treated	42.89	62.42	55.72	62.06
Abrams Creek at Mouth	treated	36.87	66.53	60.13	60.89
Abrams Creek at Vindex	treated	6.96	39.88	17.67	40.32
Abrams Creek Headwaters Right Fork	treated	13.91	46.74	41.38	44.22
<b>Emory Creek at Mouth</b>	treated	26.88	40.54	43.27	53.28
Glade Run at Mouth	treated	68.1	63.88	66.05	85.1
Laurel Run at Mouth	treated	4.15	17.09	35.11	53.7
Little Creek	treated	0.67	19.82	22.74	35.67

#### *Table 5. Summarization of index values.*



The two-way ANOVAs indicated a statistically significant effect of treatment (AMD, AMD Treated, and Reference;  $\alpha$ =0.05) on both index values in both watersheds. Year and the interaction between year and treatment type were not statistically significant predictors in either watershed with any index (Table 6).

*Table 6. Results from ANOVA run on index of biotic integrity (IBI) scores. Bolded values indicate statistically significant findings.*

P values from ANOVA on Year and Type effect on IBI scores (2017-2018)								
Watershed	Index	Year	<b>Type</b>	Year:Type				
Abrams	<b>GLIMPSS</b>	0.3944	0.0012	0.3390				
<b>Three Fork</b>	<b>GLIMPSS</b>	0.0556	$3.52E - 06$	0.5681				
<b>Abrams</b>	<b>WVSCI</b>	0.4112	0.0030	0.0601				
<b>Three Fork</b>	<b>WVSCI</b>	0.1790	1.04E-06	0.7900				

A Tukey's test was run on simple ANOVAs testing the differences between treatment types (Table 7). Results are summarized in letter groups. Statistically, significant differences in groups are represented by a different letter. Conversely, groups that share a letter are not considered significantly different. All three treatments showed statistically significant differences for both indices within Three Fork Creek. Results in Abrams Creek were less distinct and offered competing narratives depending on which index was used.

*Table 7. Tukey test results indicating which treatment types are different from one another, as denoted by letter assignments.*



Only when looking within type using paired T-tests, could we detect a statistically significant difference between years (Table 8). The treated sites were the only group to show a statistically significant difference in index values from year to year, they did so in both watersheds, with both indices for Abrams Creek - but only with GLIMPSS in Three Fork Creek.

*Table 8. Year effect on index of biotic integrity (IBI) scores. Results are separated by watershed, index, and treatment type, significant values are bolded.*



Macroinvertebrates in Three Fork Creek and Abrams Creek responded similarly to AMD remediation in each watershed. Comparisons of GLIMPSS scores show recovery toward reference conditions among treated sites (Figure 5). One-way ANOVA indicated GLIMPSS scores differ by both year (F=12.68, p<0.001) and stream type (F=17.39, p<0.001) in Three Fork Creek. Two way ANOVA (F=28.07, p<0.001) and Tukey post-tests showed a significant increase in GLIMPSS in treated streams in Three Fork Creek from pre- (2008; M=10.93, SD=9.36) to posttreatment 2017; M=35.51, SD=10.87) but did not fully reach reference conditions until 2018 (M=47.29, SD=21.98). However, 2017 and 2018 treated sites were not significantly different from 2008 reference stream conditions (M=55.54, SD=5.86). Additionally, reference streams were not statistically different from one another between the years 2017 (M=53.78, SD=6.87) and 2018 (M=64.93, SD=13.81). Raw data for macroinvertebrates in Three Fork Creek is provided in Appendices 7 through 9.



*Figure 5. Genus-level index of most probable stream status (GLIMPSS) scores pre- (2008) and post-restoration (2013-2017-2018) for the reference and treated sites of Three Fork Creek and Abrams Creek. Letters show significant differences as identified by two-way ANOVA followed by Tukey post-tests between scores of treatment types within each plot.* 

GLIMPSS scores in Abrams Creek responded very similarly to Three Fork Creek (Figure 5). Oneway ANOVA within treated sites showed GLIMPSS scores differ by year (F=16.73, p<0.001) and stream type (F=14.28, p<0.001). Two way ANOVA (F=23.9, p<0.001) and Tukey post-tests showed a significant increase in treated sites from pre- (2008; M=24.70, SD=21.78) to postrestoration (2013; M=46.87, SD=16.92), 2017(M=46.74, SD=15.32), and 2018(M=53.35, SD=14.05). GLIMPSS scores in treated sites post-restoration reached non-statistical difference from reference conditions in 2013 and continued to close the difference in 2017, and 2018 GLIMPSS scores in reference sites were not statistically different in 2008 (M=77.15, SD=NA), 2013 (M=72.79, SD=9.82), 2017 (M=70.84, SD=10.39) and 2018 (M=77.36, SD=15.14). Raw data for macroinvertebrates in Abrams Creek is provided in Appendices 3 through 6.

### Fish

Fish community was sampled at all sites during each sampling year (Appendix 1 and 2). We converted fish community data into measures of diversity using the Shannon Index for 2008, 2013 (Abrams Creek only), 2017, and 2018 sampling years at each site for each watershed with higher values representing greater fish diversity (Tables 9 and 10). Repeated measures ANOVA and subsequent Tukey post-hoc tests were used to test for significant differences between treatment types in each watershed (Figure 6). Our hypothesis was that we would see significant improvements in diversity within treated sites post-restoration.

*Table 9. Fish diversity for each site sampled in Three Fork Creek for the 2008 (pre-restoration) and 2017/2018 (postrestoration) sampling years.*



*Table 10. Fish diversity for each site sampled in Abrams Creek for the 2008 (pre-restoration), and 2013,2017,2018 (post-restoration) sampling years.*



Fish communities responded differently to restoration than macroinvertebrates in both Three Fork and Abrams Creeks. In Three Fork Creek, fish diversity was variable among treated sites postrestoration (Figure 6). Two-way ANOVA showed significant differences in diversity among years (F=3.25, p=0.048) and types (F=12.78, p<0.001). While increases in diversity occurred in treated sites, the results were nonsignificant in Tukey post-hoc tests from pre- (2008; M=0.05,SD=0.19) to post-restoration (2017; M=0.70,SD=0.86 and 2018; M=0.43,SD=0.66). Tukey tests also showed no significant differences in diversity between reference streams among years 2008 (M=0.88, SD=0.22), 2017 (M=0.98,SD=0.86), and 2018 (M=1.02,SD=0.27).



*Figure 6. Fish diversity scores for the reference and treated sites of Three Fork Creek and Abrams Creek pre- (2008) and post-restoration (2017 and 2018). Lowercase letters denote significant differences as identified by repeated measures ANOVA and Tukey post-hoc tests of scores within and between treatment types in each plot.*

Abrams Creek showed a similar increase in fish diversity post-restoration (Figure 6). In Abrams Creek, Two-way ANOVA showed significant differences in diversity between years (F=3.25, p<0.04) and by stream type (F=12.77, p<0.01). Tukey post-hoc tests show significant increases in fish diversity post-restoration in treated sites compared to pre-restoration 2008 ( $M=0$ , SD = 0), 2013 (M=0.61, SD=0.48), 2017 (M=0.79, SD=0.40), and 2018 (M=0.71, SD=0.37). Reference stream sites did not show a significant change in diversity from pre- to post-restoration 2008 (M=0.81,SD=0.64), 2013 (M=0.70, SD=0.69), 2017 (M=0.82, SD=0.60), 2018 (M=0.34, SD=0.49).

Three Fork and Abrams Creeks were severely degraded pre-restoration. It was estimated that while historically there was approximately 40 km of fishable water, only 5 km remained due to extensive pre-law mining within the Three Fork Creek watershed (Petty et al. 2008) and a total of 73 km of streams within the watershed were considered impaired pre-restoration (Pavlick et al. 2006). The goals of this stream restoration project set by the WVDEP were to improve water chemistry and aesthetics in order to increase recreational use while also restoring macroinvertebrate and fish communities (WVDEP 2013).

Fish diversity is greatly improved throughout the Three Fork Creek watershed. Prior to restoration, fish were only found in reference sites not impacted by acid mine drainage, and many considered the main stem of Three Fork Creek to be "dead." Fish diversity has improved in treated sites as reference stream sites are now connected to more of the watershed, and water quality allows fish movement in and out of the watershed. However, within treated sites, there is high variability in diversity. Many impacted tributaries and upstream main stem sites have no fish or are dominated by one or two tolerant species (e.g. creek chub, *Semotolis atromaculatus*, and blacknose dace, *Rhinichthys atratulus*). As water quality improves downstream, more species were found within our sites (Appendix 1).

Like Three Fork Creek, Abrams Creek is also showing great improvements in fish diversity across treated sites, and treated sites are similar to reference diversity scores. Variability, however, is less in Abrams Creek. Only one site in Abrams Creek has a diversity score of zero compared to multiple sites in Three Fork Creek. A few explanations of this discrepancy could be:

- 1) Abrams Creek is less than half the size of Three Fork Creek so the distance for source populations to travel to occupy treated sites may be less (Lorenz and Feld 2013);
- 2) The state of the regional species pool could be different between the two watersheds depending on the regional condition surrounding the watersheds (Martin 2010; Merriam and Petty 2016); or
- 3) The volume of AMD in some locations just may be so extensive that in-stream treatment is not effective enough to restore a functioning fishery (Freund and Petty 2007).

The effect of regional species pools on the ability of restored streams to be repopulated has become more explored recently (Martin 2010; Merriam and Petty 2016) and may be an interesting factor to further explore. Additionally, it is important to note improvements of diversity do not indicate species' abundances which is likely the discrepancy in our results between the apparent diversity improvements and the failure to recover functional fisheries with EUs.

### Water chemistry

Principal components analysis (PCA) was used to characterize the dominant patterns of variation within the water chemistry dataset. Prior to analysis, all chemicals except for pH and specific conductance were log+1 transformed. Total acidity was removed from the analysis due to its correlation with other elements in the analysis (Merovich et al. 2007). Cadmium, chromium, and selenium were removed from the analysis due to all water samples being below the detection limit for these elements. Any concentrations below the detection limit for other elements were assumed to have a concentration of zero for that element. Significant principal components were chosen using a broken stick analysis where principal components are retained when their corresponding eigenvalues are greater than their predicted broken stick values (Jackson 1993). Samples were grouped in ordination space by their stream type (reference, AMD, or treated). Non-metric analysis of variance using distance matrices (ADONIS) followed by pairwise permutation multivariate analysis of variance (MANOVA) was used to determine if PCA results differed between stream types. All PCA and ADONIS analyses were completed in the package vegan (Oksanen et al. 2013) in R (R Core team 2017). AMD sites for Abrams Creek were not included in pairwise comparisons due to only having one AMD site to include in the analysis.

In Abrams Creek, broken stick analysis resulted in two PCs which together accounted for 56.9% (2008), 69.9% (2013), 61.1% (2017), and 59.8% (2018) of the variance in water chemistry (Table 11; Figure 7). Principle component correlations greater than 0.4 in either direction can be found for all years and correspond with scatter plots of each PCA (Table 11; Figure 7). Variables not correlated with either PC1 or PC2 to a strength of 0.4 were omitted from Table 11. Global ADONIS showed that water chemistry differs by stream type (F=35.25, p<0.01), and pairwise permutation MANOVAs showed significant differences in water chemistry signatures between reference and treated sites (p<0.01).

PCA analysis for Three Fork Creek and Abrams Creek were similar to well-defined water chemistry signatures for each site type. In Three Fork Creek, broken stick analysis resulted in two principal components (PCs), which together accounted for 64.9% (2008), 74.4% (2017), and 72.4% (2018) of the variance in water chemistry. Principle component correlations greater than 0.4 in either direction can be found for all years and correspond with scatter plots of each PCA (Table 11; Figure 8). Variables not correlated with either PC1 or PC2 to a strength of 0.4 were omitted from the list. ADONIS showed water chemistry differs by treatment type (F=21.56, p<0.01), and pairwise permutation MANOVAs show that each stream type has a significantly different water signature from other stream types (all comparisons, p< 0.01; Figure 8).

*Table 11. LEFT: Pearson correlations of variables to principle component 1 and principle component 2 for Abrams Creek. +/- indicates the direction of correlation as relates to Figure 7. RIGHT: Pearson correlations of variables to principle component 1 and principle component 2 for Three Fork Creek. +/- indicates the direction of correlation as relates to Figure 8.*







*Figure 7. Scatter plots of principal components (PC) 1 and 2 scores for every water chemistry sample in Abrams Creek. Points are colored and shaped by site types (red=AMD, green=reference, blue=treated).* 



*Figure 8. Scatter plots of principal components (PC) 1 and 2 scores for every water chemistry sample in Three Fork Creek. Points are colored and shaped by site types (red=AMD, green=reference, blue=treated).* 

Water chemistry results indicate significant improvements in post-restoration in treated sites within the watersheds. Although still intermediary between AMD impaired and reference conditions, alkalinity and pH increases with consequent decreases in heavy metals have pushed chemical conditions toward reference conditions. Some chemicals, like sulfate and magnesium, are extremely difficult to reduce with this type of AMD remediation (i.e. in-stream treatment) and are still elevated post-treatment in Abrams Creek (Table 12) and Three Fork Creek (Table 13; Freund and Petty 2007). This, combined with increased levels of TSS and Na, highly correlated with PC2 (Figure 4), are the likely reasons for the separation of chemical signatures between reference and treated sites. Due to the extent of impairment, our results show it is unlikely that water chemistry will ever fully reflect unimpaired reference conditions with this type of in-stream treatment. Even so, it should be noted that the water quality improvements throughout the watersheds are extensive and have not only improved conditions within the creeks themselves but likely the rivers to which they drain.



*Table 12. Chemical summary table of water chemistry samples in Abrams Creek treated sites pre- (2008) and postrestoration (2013, 2017, and 2018). Mean concentrations and standard deviations (SD) of each selected chemical parameter among all treated sites are listed.*

*Table 13. Chemical summary table of water chemistry samples in Three Fork Creek treated sites pre- (2008) and post-restoration (2017 and 2018). Mean concentrations and standard deviations (SD) of each selected chemical parameter among all treated sites are listed.*



Macroinvertebrate communities are showing large improvements post-restoration in both watersheds, and their responses are very similar. Both watersheds are showing a significant increase of GLIMPSS in treated sites post-restoration, which are approaching reference conditions. This is reflected in how both watersheds show that while treated sites were significantly different than reference sites in 2017, they were not different than reference sites from 2008. This suggests the improvement of the macroinvertebrate communities in the treated sites could be inflating the communities at the watershed scale even though reference scores were not statistically different from pre- to post-restoration.

Additionally, GLIMPSS scores in both watersheds show that reference and treated sites have significantly different scores in 2017. This difference in the genus level measure suggests that although improvements are being made functionally and we are seeing increases in EPT values, certain taxa may not be returning to these sites. McClurg et al. (2007) found that in treated acidic streams, there was considerable variation in benthic macroinvertebrate communities based on the distance to treatment. Mixing zones that were less than 2 km downstream of treatment caused highly variable water quality and benthic macroinvertebrate communities, which more closely resembled untreated acidic sites rather than unimpacted reference sites. Site locations within Three Fork Creek and Abrams Creek vary throughout the watersheds in proximity to treatment, and the variability in community structures may be reflected in this relationship. Additionally, if treated and reference streams are isolated within the watershed, it is possible that a surrounding metacommunity with a deflated species pool could be affecting the ability of certain taxa to reach these streams (Merriam and Petty 2016).

## Objective 2 – Quantify the long-term success of watershed-scale restoration efforts within Abrams Creek and Three Fork Creek by comparing observed and predicted response.

EUs in Three Fork Creek reached predictions for macroinvertebrate diversity but was below predictions for all fishery EUs (Figure 9). Kolmogorov–Smirnov tests showed post-restoration diversity EUs did not accumulate at a different rate than predicted EUs for post-restoration (D=0.08, p=0.27) but also for pre-restoration EUs (D=0.09, p=0.17). Overall, fishery EUs were much lower post-restoration than predicted but still accumulated more quickly than pre-restoration EUs (D=0.56, p<0.01). Similarly, cold-water fishery EUs and warm-water fishery EUs were much lower than predicted post-restoration but accumulated more quickly than pre-restoration EUs for cold-water (D=0.60, p<0.01) and warm-water fisheries (D=0.58, p<0.01). In total, 68% of historical diversity EUs and 19% of historical overall fishery EUs have been recovered. Overall, fishery EUs were 84% lower than predicted post-restoration.



*Figure 9. Ecological Units in Three Fork Creek for diversity, overall fishery, cold-water fishery, and warm-water*  fishery. Diversity units in this figure represent macroinvertebrate diversity. Overall fishery units represent a *combination of trout and warm-water fisheries.*

Like other metrics, Abrams Creek EUs are responding similarly to Three Fork Creek EUs (Figure 10). Diversity EUs increased from 14.49 cumulative miles pre-restoration to 25.76 cumulative miles post-restoration. Historically Abrams Creeks had 34.66 functioning miles for macroinvertebrate diversity, so 74% of historic stream miles are now functional post-restoration as compared to 42% before the restoration project. Kolmogorov–Smirnov tests show significant increases in the accumulation of diversity units from pre- and post-restoration  $(D=0.42, p<0.01)$ . Additionally, post-restoration EUs in 2017 did not accumulate differently than what was predicted post-restoration (D=0.18, p=0.65). Like Three Fork Creek, Abrams Creek fishery EUs did not recover to predicted EUs after restoration. Historically, it was expected that 34.66 cumulative miles of EUs existed in the watershed before any mining activity. It was degraded to 10.35 miles pre-restoration and restored to only 14.97 miles in 2017. Kolmogorov–Smirnov tests show that EUs did not accumulate at predicted rates post-restoration (D=0.36, p=0.03) nor were they different than pre-restoration (D=0.30, p=0.10). When split into stocked and brook trout EUs, stocked trout increased post-restoration ( $D=0.52$ ,  $p<0.01$ ) from pre-restoration EUs, but was much lower than predicted EUs (D=0.36, p=0.02). Stocked trout EUs increased by 33% postrestoration but only reached 47% of historical values. Brook trout EUs did not increase from prerestoration EUs (D=0.24, p=0.29) so did not reach predicted EUs (D=0.52, p<0.01). Only 8.90 of historical 30.72 brook trout EUs were calculated to be functional post-restoration.



*Figure 10. Ecological Units (EU) in Abrams Creek for diversity, brook trout, stocked trout, and overall fishery. Diversity units in this figure represent macroinvertebrate diversity. Overall, fishery units represent a combination of native brook trout and stocked trout.*

Fishery related EUs for Three Fork Creek and Abrams Creek came up short of predictions for the restoration projects. Our fish diversity results show diversity scores significantly improving and reaching our reference conditions in both watersheds, but EUs indicate we are not achieving a functioning fishery. This indicates that although diversity is improving, fisheries are not developing, which could indicate a density issue. It was predicted that the majority of the historic warm-water fishery would be recovered in Three Fork Creek post-restoration, and the warm-water fishery was predicted to have a better recovery than the cold-water fishery due to warm-water habitats being far enough downstream of treatment to have stable conditions for fish. Our findings show that the fishery has not had much of recovery when compared to pre-restoration with warmwater EUs only accumulating 1.3 additional functioning stream miles post-restoration. Similarly, Abrams Creek only recovered 0.3 miles of brook trout EUs and 4.2 miles of stocked trout EUs. These findings alone tell us that although these restoration projects successfully recovered macroinvertebrate (i.e. WVSCI), EUs, and fish diversity has increased, it was not able to recover a functional fishery within the watershed. It will likely require out-of-stream (i.e. at source) treatment to recover functioning fisheries in watersheds with this extensive of impairment.

## Objective 3 – Develop a remediation prioritization framework for AMDimpacted systems that simultaneously maximize ecological and socioeconomic benefits.

Our data for our model (Figure 11) came from the table shown below and is also available digitally at the link below for download and viewing.



*Figure 11. Benefit transfer values used in study.*

https://drive.google.com/file/d/1KHbUnuXfUoVkbBLtjHSIWkhZ5Z-YR2Cy/view?usp=sharing

The benefit transfer approach utilizes nonmarket valuation data from existing studies called study sites to value natural resources at sites of interest, called a policy site (Rosenberger and Loomis 2001; Bergstorm and Taylor 2006). In order to implement benefit transfer, a metaregression model was estimated using data from the study sites to formulate a statistical relationship between the reported study site values and study site characteristics. By applying policy site data to the metaregression model, a nonmarket value was derived for the sites.

From our literature review, the variables we have been assembling include the value of ecosystem services, amount of damage, drainage area, length of the stream, age composition, county, and state fixed effects, type of study (travel cost, hedonic price model, contingent valuation), and ecosystem services.

By applying policy site data to the meta-regression model, a nonmarket value could be derived for the watershed study sites.

The meta-analytical regression model is:

$$
ln y_{ij} = \gamma E S_{ij} + \beta^b X_{ij}^b + \beta^s X_{ij}^s + \beta^c X_{ij}^c + \varepsilon_{ij}
$$

*where:*

- *ES* includes the ecosystem services provided by the stream (with potential interactions across ecosystem services),
- $\bullet$   $\chi$ <sup>b</sup> is a vector describing the water body characteristics (i.e. type of water body, size of the water body),
- $\bullet\quad$   $X^s$  is a vector describing the study characteristics (i.e., survey method, payment vehicle, elicitation format) and
- *Xc* includes context-specific explanatory variables.
- In the equation, the subscript *i* takes values from 1 to the number of studies, and subscript *j* takes values from 1 to the number of observations,  $\varepsilon$  is the usual error term and the vectors  $\beta^b$ ,  $\beta^s$ ,  $\beta^c$  and *γ* contain coefficients to be estimated for the explanatory variables in  $X^b$ ,  $X^s$ ,  $X^c$ , and ES, respectively.

The identification and screening were done by keyword searches from the literature search database for ecosystem services, acid mine drainage, restoration, and benefit transfer. (https://drive.google.com/file/d/1KHbUnuXfUoVkbBLtjHSIWkhZ5Z-YR2Cy/view?usp=sharing) Variable extraction and selection resulted in relevant variables for the meta-analysis and construction of the benefit transfer function. Results indicated the benefit of acid mine remediation is expected to render a benefit of \$14.44 per day per person on an average across this region, while an average willingness to pay for fishing ranges from \$6 to \$95 for restoration.

# **Conclusion**

Future research on these watersheds should focus on the discrepancy between increased fish diversity to reference conditions and the continued lack of functional fisheries. Density metrics that explore species abundances as well as documenting the presence/absence of certain species or functional groups of fish may help explain why fisheries in both watersheds are not fully recovering. This data will be needed if managers plan to improve the fisheries of Abrams and Three Fork Creeks. Additionally, further macroinvertebrate analysis can give valuable insight into what taxa are not returning to our treated locations and whether or not functional diversity in our treated locations is comparable to our reference sites. This information could help managers predict the ability of AMD restoration to repopulate sensitive taxa.

Historically, stream restoration projects have focused on the reach-scale without concern of watershed or regional-scale processes that surround them. Multiple studies have found these site-specific, reach scale approaches are not seeing the biological uplift expected due to the strict focus of habitat improvement rather than the reconnection of isolated populations or not fully addressing the sources of impairment at the watershed-scale (McClurg et al. 2007 ; Palmer et al. 2014). In a study of fish populations in the central Appalachians, Martin (2010) found that local biological conditions were independent of local conditions for stream fishes. This finding supports that site-specific restorations are not going to find the biological improvements only by improving habitat or water quality at the local scale. Further, with macroinvertebrates, although they are usually a good indicator of local water quality (Freund and Petty 2007), regional processes (i.e. dispersal) dictate the communities that will reside there (Merriam and Petty 2016).

Our results clearly show the benefits of focusing restorations at the watershed-scale. Some biological and chemical attributes are still not fully recovered in either watershed, but that is expected due to the severity of impairment which AMD causes and the infeasibility of complete restoration in these systems. Still, macroinvertebrate indices and fish diversity were greatly improved in both Abrams Creek and Three Fork Creek due to the improved water chemistry and reconnection of isolated tributaries that serve as sources to repopulate the watersheds. Our results indicate that restoration projects which focus on the watershed scale to improve connections to good conditions both locally and regionally are expected to be more successful than improving local conditions alone. Our results, especially at Abrams Creek, also show that it may take more extended timeframes to see the full benefits of restoration and for biological communities to fully recover.

Regional impairment beyond the watershed may also affect the ability of a stream to recover. Three Fork Creek is showing great improvements for macroinvertebrates throughout our restored reaches, but fish are still struggling to repopulate the watershed due to high regional impairment and a blockage to the fish movement (i.e. Tygart Lake dam). Although this restoration project was focused mainly on improving water quality before reaching the Tygart River, future restoration projects may need to focus regionally to reconnect healthy watersheds to restored areas. Additionally, the differences in recovery between macroinvertebrates and fish in our watersheds suggest that monitoring both fish and macroinvertebrates should be a part of ecological monitoring programs. Macroinvertebrates can give managers a good idea of local conditions, but fish may be able to tell a larger story of regional conditions both within the watershed and beyond.

Our findings from the ANOVA show that when all three treatments are taken into account together, a year is not a significant factor. This can be seen in the T-test results as well with AMD and reference streams not varying significantly between years. However, the AMD remediated group (except TF WVSCI) had significant differences between 2017-2018. This supports the idea that the remediation efforts are still in the process of improving as every metric in treated sites improved between 2017-2018.

Our results can help watershed managers by showing that long-term monitoring and regionalscale thinking can help to improve restoration projects. With the completed cycle of our adaptive management framework, our results can be used to address successes, shortcomings, and where changes can be made to continue to improve the ecological condition of these watersheds. Our results also show that even with huge improvements, full recovery of macroinvertebrate and fisheries to reference conditions of systems highly degraded by AMD may not be possible without at-source treatment. Even so, our results show that watershed-scale restoration leads to many ecological improvements, and regional-scale processes play a large role in ecosystem recovery.

Some of the issues with benefit transfer approach for ecosystem service calculations have been noted by King et al. (2000) and include: Benefit transfer may not be accurate, except for making gross estimates of recreational values, unless the sites share all of the site, location, and user-specific characteristics; good studies for the policy or issue in question may not be available; it may be difficult to track down appropriate studies since many are not published;



*Figure 12. Summary of ecosystem service benefits from AMD versus annual doser costs (Hansen et al. 2010).*

reporting of existing studies may be inadequate to make the needed adjustments; adequacy of existing studies may be difficult to assess; extrapolation beyond the range of characteristics of the initial study is not recommended; benefit transfers can only be as accurate as the initial value estimate; and; unit value estimates can quickly become dated. However, even with these limitations, the method is typically less costly than conducting an original valuation study, and the economic benefits can be estimated more quickly than when undertaking an original valuation study. It also can be used as a screening technique to determine if a more detailed, original valuation study should be conducted, and the method can easily and quickly be applied for making gross estimates of recreational values. The most important consideration is that the more similar the sites and the recreational experiences, the fewer biases will result (King et al. 2000).

The results of this study for estimating the benefits of the restoration of acid mine drainage adds to the previous work. Specifically, Hansen et al. (2010) found that AMD remediation can benefit local economies. They used a variety of techniques to estimate the doser costs versus local economic benefits from both direct spending and indirect as well as willingness-to-pay, which is a survey technique. A summary graphic from their report is shown above (Figure 12).

This highlights the ecosystem service benefits from AMD remediation compared to annual doser costs. While our study did not find the direct impact of money spent in the watershed, a transferable value was attained to better understand the benefits of restoration. Future work and research are needed to better understand the more remote values placed on smaller streams such as Three Fork Creek and Abrams Creek that do not have an established whitewater recreation industry or access to support increased fishing days opportunities.
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## **Appendices**

**Appendix 1.** Fish species counts by site in Abrams Creek watershed for all sampling years (2008, 2013, 2017, 2018). NS = Not sampled. AMNE = *Ameiurus nebulosus* (brown bullhead), AMRU = *Amboplites rupestris* (rock bass), CAAN = *Campostoma anomalum* (central stoneroller), CACO = *Catostomus commersoni* (white sucker), COCA = *Cottus caeruleomentum* (blue ridge sculpin), ETFL = *Etheostoma flabellare* (fantail darter), HYNI = *Hypentelium nigricans* (northern hogsucker), LECY = *Lepomis cyanellus* (green sunfish), LEMA = *Lepomis macrochirus* (bluegill), MIDO = *Micropterus dolomieu* (smallmouth bass), MIPU = *Micropterus punctulatus* (spotted bass), ONMY = *Oncorhynchus mykiss* (rainbow trout), RHAT *= Rhinichthys atratulus* (blacknose dace), SAFO = *Salvelinus fontinalis* (brook trout), SEAT = *Semotilus atromaculatus* (creek chub).





**Appendix 2.** Fish species counts by site in Three Fork watershed for all sampling years (2008, 2017, 2018). NS = Not sampled. AMNA = *Ameiurus natalis* (yellow bullhead). AMNE = *Ameiurus nebulosus* (brown bullhead), AMRU = *Amboplites rupestris* (rock bass), CAAN = *Campostoma anomalum* (central stoneroller), CACO = *Catostomus commersoni* (white sucker), COBA = *Cottus bairdii* (mottled sculpin), ETFL = *Etheostoma flabellare* (fantail darter), HYNI = *Hypentelium nigricans* (northern hogsucker), LECY = *Lepomis cyanellus* (green sunfish), LEGI = *Lepomis gibbosus* (pumpkinseed sunfish) LEMA = *Lepomis macrochirus* (bluegill), MIDO = *Micropterus dolomieu* (smallmouth bass), MIPU = *Micropterus punctulatus* (spotted bass), MISA = *Micropterus salmoides* (largemouth bass), NOMI = *Nocomis micropogon* (river chub), NORU *= Notropis rubellus* (rosyface shiner), PECA = *Percina caprodes* (logperch), PINO = *Pimephales notatus* (bluntnose minnow), RHOB = *Rhinichthys obtusus* (blacknose dace), SAFO = *Salvelinus fontinalis* (brook trout), SATR = *Salmo trutta* (brown trout), SAVI = *Sander vitreus* (walleye), SEAT = *Semotilus atromaculatus* (creek chub).





	Abrams Creek at Mouth	Emory Creek	Abrams Creek above Emory	at Laytons Abrams Creek	Johnnycake at Mouth	Abrams Creek above Johnnycake	Glade Run	Abrams Creek above Glade	Laurel Run	Abrams Creek above Laurel	Abrams Creek at Vindex	42 $\mathfrak{S}$ $\vec{a}$ eek Ğ Abrams	Б Abram Creek HW	Abram Creek HW RF
Year	2008	2008	2008	2008	2008	2008	2008	2008	2008	2008	2008	2008	2008	2008
Oligochaeta	4	4	5	$\mathbf{1}$	22	1	12	$\overline{2}$	$\mathbf 0$	0	$\overline{2}$	2 <sup>1</sup>	9	4
Planorbidae	0	$\mathbf 0$	0	0	$\mathbf 0$	$\mathbf 0$	$\pmb{0}$	$\mathbf 0$	0	0	0	$\mathbf{1}$	0	$\mathbf 0$
Lymnaeidae	0	$\mathbf 0$	0	0	$\mathbf 0$	$\mathsf 0$	$\pmb{0}$	0	0	0	$\pmb{0}$	$\mathbf 1$	0	$\mathbf 0$
Snails(UNK)	$\mathbf 0$	$\mathbf 0$	0	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf 0$	0	0	0	$\mathbf{1}$	0	$\mathbf 0$
Gammarus	0	$\mathbf 0$	0	$\mathbf 0$	$\mathbf 0$	$\pmb{0}$	$\mathbf 0$	$\mathbf{0}$	0	0	0	0	0	$\mathbf 0$
Caecidotea	$\mathbf{1}$	$\mathbf 0$	0	0	$\mathbf 0$	$\mathbf 0$	$\pmb{0}$	$\mathbf{1}$	0	0	0	0	$\mathbf 0$	$\mathbf 0$
Baetidae(UNK)	$\overline{2}$	0	5	6	3	$\mathbf 0$	$6 \overline{6}$	$\mathbf 0$	0	0	1	0	0	$\mathbf 0$
Accentrella	$\overline{2}$	$\mathbf 0$	3	3	5	$\mathbf{0}$	5	$\mathbf 0$	0	0	0	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$
<b>Baetis</b>	$\mathbf 0$	$\mathbf 0$	$\overline{2}$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	17	$\mathbf{0}$	0	0	0	$\mathbf 0$	0	$\mathbf 0$
Plauditus	67	$\mathbf 0$	58	112	$\overline{7}$	$\mathbf 0$	$\pmb{0}$	0	0	0	$\pmb{0}$	0	$\mathbf 0$	$\mathbf 0$
Heterocloeon	$\mathsf 0$	0	0	0	$\mathbf 0$	$\mathsf 0$	$\pmb{0}$	$\mathsf 0$	0	0	0	0	0	$\mathbf 0$
Centroptilum	0	$\mathbf 0$	0	0	0	$\mathsf 0$	$\pmb{0}$	0	0	0	0	0	0	$\mathbf 0$
Heptageniidae(UNK)	$\mathbf 0$	$\mathbf 0$	$\mathbf{1}$	$\mathbf 0$	$\overline{2}$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	0	0	0	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$
Epeorus	0	$\mathbf 0$	0	0	5	$\mathbf 0$	$\pmb{0}$	$\mathbf 0$	0	0	0	0	$\mathbf 0$	$\mathbf 0$
Heptagenia	0	$\mathbf 0$	0	0	0	$\mathbf 0$	$\mathbf{0}$	0	0	0	0	0	$\mathbf 0$	$\mathbf 0$
Stenonema/Maccaffertium	0	0	0	0	3	$\mathbf 0$	$\mathbf{1}$	$\mathbf 0$	0	0	0	0	0	$\mathbf 0$
Cinygmula	0	$\mathbf 0$	0	0	$\mathbf 0$	0	$\mathbf{1}$	$\mathbf 0$	0	0	0	$\mathbf 0$	0	0
Stenacron	0	0	0	$\mathbf 0$	$\mathbf 0$	$\pmb{0}$	$\pmb{0}$	0	0	0	0	$\mathbf 0$	0	$\mathbf 0$
Isonychia	0	0	0	0	0	$\pmb{0}$	$\pmb{0}$	0	0	0	$\pmb{0}$	0	$\mathbf 0$	$\mathbf 0$
Leptophelbiidae(UNK)	0	$\mathbf 0$	0	$\mathbf 0$	$\mathbf{0}$	$\mathsf 0$	$\mathsf 0$	0	0	0	$\pmb{0}$	0	$\mathbf 0$	$\mathbf 0$

**Appendix 3.** Benthic macroinvertebrate data for Abrams Creek by sample location in 2008. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.











**Appendix 4.** Benthic macroinvertebrate data for Abrams Creek by sample location in 2013. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.













Site	Laytons ම Creek Abram	Mouth ල Abrams	Emory ω ੋ 욶 Abram	eek δ mo Em	Laurel $\mathbf \omega$ š 윿 Abram	Laurel	Glade above Abram	Glade	ڀ م त्त ≏	Vindex ල Abram	$\overline{a}$ $^\copyright$ Abram	Creek Little	Right Abram HW	(EM2) 2 ≧ ⊾ ě ბ Emory	$\sim$ 요 ξ	EM1) mor шī	Run Johnnycake Upper.	mouth ම $\blacksquare$ 교 cake Johnny
Year	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017
Oligochaeta	$\overline{0}$	0	$\overline{0}$	$\overline{0}$	$\mathbf{0}$	$\overline{0}$	$\overline{0}$	$\mathbf{0}$	$\overline{0}$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\overline{0}$	$\overline{0}$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$
Planorbidae	$\overline{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	0	$\mathbf{0}$	0	$\overline{0}$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	0	$\overline{0}$	$\mathbf 0$
Lymnaeidae	$\overline{0}$	0	0	$\mathbf{0}$	0	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	0	0	0	$\overline{0}$	$\mathbf 0$	$\overline{0}$	0	$\Omega$	$\mathbf 0$	$\mathbf 0$
Snails(UNK)	$\overline{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	0	0	$\overline{0}$	$\mathbf 0$	0	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf 0$
Gammarus	$\mathbf{0}$		0	$\Omega$	$\Omega$	$\mathbf 0$	$\mathbf 0$	$\Omega$	0		$\Omega$	$\mathbf 0$	$\mathbf 0$	0	21	$\Omega$	$\Omega$	$\mathbf 0$
Caecidotea	1	O	$\Omega$	$\Omega$	$\Omega$	$\mathbf{0}$	$\mathbf 0$	$\Omega$	O	$\Omega$	$\Omega$	8	$\mathbf 0$	$\Omega$	$\mathbf{0}$		$\Omega$	$\mathbf 0$
Baetidae(UNK)	$\overline{0}$	$\Omega$	0	$\mathbf{0}$	$\Omega$	$\mathbf{0}$	$\mathbf 0$		0	0	$\Omega$	$\overline{0}$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\overline{0}$
Accentrella	$\overline{3}$	22			3	5	$\overline{3}$	6	7	0		$\mathbf{0}$	$\overline{2}$		$\Omega$	3	0	8
Baetis	$\mathbf{1}$	4		1	$\Omega$	4	$\overline{4}$	44	2	$\Omega$		$\overline{0}$	$\mathbf 0$	82	38	26	16	10
Plauditus	72	42	56	18	123	$\mathbf{0}$	109		122	3	$\overline{2}$	$\overline{0}$	5	$\overline{2}$	3	37	10	48
Heterocloeon	$\overline{0}$	6	0	$\mathbf 0$	3	$\mathbf{0}$	$\vert$ 1	$\Omega$	1	0	0	$\overline{0}$	$\mathbf 0$	$\overline{0}$	0	$\Omega$	$\overline{0}$	$\mathbf 0$
Centroptilum	$\mathbf{0}$	0	$\Omega$	$\Omega$	0	$\mathbf{0}$	$\mathbf{0}$	$\Omega$		$\Omega$	$\Omega$	$\mathbf 0$	$\pmb{0}$	0	$\Omega$		$\Omega$	$\mathbf 0$
Heptageniidae(UNK)	$\overline{0}$	$\Omega$	0	$\overline{0}$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\Omega$	$\mathbf{0}$	0	0	$\overline{0}$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	O	0	$\mathbf 0$
Epeorus	1	0		$\Omega$	0	$\mathbf 0$	$\mathbf{0}$	2	0	$\Omega$	0	$\mathbf 0$	$\pmb{0}$	6	9		$\Omega$	$\overline{4}$
Heptagenia	$\mathbf{0}$	$\overline{0}$	$\Omega$	$\overline{0}$	$\mathbf{0}$	$\mathbf{0}$	$\overline{0}$	$\Omega$	$\mathbf{0}$	$\Omega$	0	$\overline{0}$	$\pmb{0}$	$\overline{0}$	$\mathbf{0}$	O	$\mathbf{0}$	$\mathbf 0$
Stenonema/Maccaffertium	$\mathbf{0}$			$\Omega$	$\Omega$	$\mathbf{0}$	$\mathsf{O}\xspace$		0	$\Omega$	$\mathbf 0$	$\mathbf{0}$	$\pmb{0}$	0	$\Omega$		$\Omega$	5 <sup>5</sup>
Cinygmula	$\mathbf{0}$	0	$\Omega$	$\overline{0}$	$\mathbf{0}$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	0	0	0	$\overline{0}$	$\pmb{0}$	$\mathbf{0}$	$\mathbf{0}$	<sup>0</sup>	$\Omega$	$\mathbf 0$
Stenacron	2	0	0	$\mathbf{0}$	0	$\mathbf{0}$	$\mathbf{0}$		0	$\Omega$	0	$\overline{0}$	$\pmb{0}$	$\overline{0}$	2		3	$\overline{3}$
Isonychia	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	0	$\Omega$	0	$\overline{0}$	$\pmb{0}$	$\mathbf{0}$	$\Omega$	0	0	$\mathbf 0$
Leptophelbiidae(UNK)	$\mathbf{0}$	0	0	0	0	$\mathbf{0}$	$\mathbf 0$	$\Omega$	0	$\Omega$	$\Omega$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	0	$\Omega$	0	0
Paraleptophlebia	$\overline{0}$	0		$\overline{0}$	$\overline{0}$	$\mathbf{0}$	$\mathbf{0}$	6	$\mathbf{1}$	$\mathbf{0}$	$\Omega$	$\overline{0}$	$\mathbf 0$	10	28	$\overline{9}$	13	9

**Appendix 5.** Benthic macroinvertebrate data for Abrams Creek by sample location in 2017. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.











Site	Laytons ම Abram Creek	Mouth ☺ Abram	Emory above m Abr:	Creek Emory	Laurel Abram above	laurel	above Glade Abram	Glade	above JC Creek Abram	Vindex ☺ Abram	$\overline{a}$ $^{\circ}$ Abram	Creek Little	Right HWs Abram	$\sim$ $\geqq$ Creek Emory	$\sim$ Emory š	$\blacksquare$ $\geq$ Emory	Run Jonnycake ă <b>B</b>	Mouth Jonnycake	$\overline{ }$ Emory š
Year	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018	2018
Accentrella	29	23	28	5 <sup>1</sup>	46	$\mathbf{1}$	101	18	71	16	$\overline{0}$	$\mathbf{0}$	43	$\vert$ 1	12	$\overline{0}$	15	$\overline{3}$	$\mathbf{1}$
Acroneuria	$\overline{3}$	$\vert$ 1	3	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	1	$\overline{0}$	$\mathbf{1}$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	12	$\overline{0}$	6	$\overline{0}$
Aeshna	$\Omega$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\Omega$	0	$\overline{0}$	$\mathbf 0$	$\Omega$	$\mathbf{0}$	$\overline{0}$
Aeshnidae (UNK)	$\Omega$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	$\overline{0}$
Agapetus	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	0	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	$\overline{0}$
Agnetina	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	0	$\overline{0}$	0	$\overline{0}$		$\mathbf{0}$
Allocapnia	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\overline{0}$	0	$\overline{0}$	$\Omega$	$\mathbf{0}$
Alloperla	$\Omega$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\overline{2}$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	0	$\overline{0}$	$\mathbf{1}$	$\mathbf{1}$	<sup>n</sup>	$\overline{0}$
Ameletus	$\Omega$	1	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	1	$\Omega$	0	n	$\overline{0}$
Amphinemura	$\Omega$	6	$\mathsf 0$	11	12	12	6	2	$\overline{2}$	$\mathbf{1}$	$\overline{2}$	$\mathbf{0}$	51	3	$\overline{3}$	$\mathbf{1}$	$\Omega$	$\Omega$	11
Anchytarsus	$\Omega$	$\Omega$	$\mathsf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	$\mathbf{1}$
Ancyronyx	$\Omega$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	0	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\overline{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$
Antocha	$\Omega$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\overline{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$
Asellidae(UNK)	$\Omega$	$\mathbf{0}$	$\mathsf 0$	$\mathbf 0$	$\overline{7}$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	11	$\Omega$	$\mathbf{1}$	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	2
Baetidae(UNK)	$\overline{2}$	$\Omega$	$\overline{2}$	3	$\overline{2}$	$\mathbf{1}$	$\mathbf{0}$	$\mathbf{0}$	4	$\mathbf 0$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\Omega$	1	$\Omega$	$\mathbf{0}$
Baetis	17	26	$\overline{7}$	$\overline{2}$	37	$\overline{2}$	27	33	33	4	$\overline{3}$	$\mathbf{1}$	3	$\mathbf{0}$	42	50	16	43	$\mathbf{1}$
Beloneuria	$\Omega$	$\mathbf{0}$	$\mathsf 0$	$\mathbf 0$	$\mathsf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathsf 0$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf{0}$	U	$\mathbf{0}$	$\overline{0}$	$\Omega$	$\mathbf 0$	$\Omega$	$\mathbf{0}$
<b>Bezzia</b>	$\Omega$	$\mathbf 0$	$\mathbf 0$	$\mathbf{1}$	0	$\mathbf 0$	$\mathbf{1}$	$\mathbf{0}$	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	2	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\Omega$	$\mathbf{0}$
Blepharicera	$\Omega$	$\mathbf{0}$	0	$\mathbf 0$	0	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf 0$	$\overline{0}$	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$
Boyeria	$\Omega$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	0	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	$\Omega$	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	$\mathbf{0}$
Brachycentrus	$\Omega$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\mathsf 0$	$\mathsf 0$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\Omega$	$\Omega$	$\Omega$	$\mathbf 0$
Caecidotea	$\Omega$	$\Omega$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{0}$	$\Omega$	0	$\mathbf{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$	$\Omega$	$\Omega$	n	$\mathbf 0$
Cambarus	$\mathbf{1}$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\overline{2}$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf{1}$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	1	$\Omega$	$\overline{2}$	$\overline{2}$	3		$\mathbf{1}$
Centroptilum	$\Omega$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	$\mathbf{0}$
Ceratopogonidae(UNK)	$\overline{0}$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf{1}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\overline{0}$	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\Omega$	$\Omega$	$\Omega$	$\overline{0}$
Ceratopsyche	$\Omega$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\overline{0}$	$\Omega$	$\mathbf{0}$	$\overline{0}$	$\mathbf{0}$	$\Omega$	$\Omega$	$\mathbf{0}$
Cernotina	$\Omega$	$\mathbf 0$	$\mathbf{0}$	$\mathbf 0$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\overline{0}$	$\overline{0}$	$\Omega$	$\overline{0}$	$\overline{0}$	$\mathbf 0$	$\Omega$	$\Omega$	$\overline{0}$
Chelifera	$\Omega$	0	$\mathbf 0$	$\Omega$	$\mathbf 0$	$\mathbf 0$	$\overline{0}$	$\mathbf{0}$	$\mathbf 0$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf{0}$	$\mathbf 0$	$\overline{0}$	$\overline{0}$	$\mathbf{1}$	$\overline{0}$		$\overline{0}$

**Appendix 6.** Benthic macroinvertebrate data for Abrams Creek by sample location in 2018. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.















**Appendix 7.** Benthic macroinvertebrate data for Three Fork Creek by sample location in 2008. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.


















## **Appendix 8.** Benthic macroinvertebrate data for Three Fork Creek by sample location in 2017. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.

























**Appendix 9.** Benthic macroinvertebrate data for Three Fork Creek by sample location in 2018. With the exception of Chironomidae, macroinvertebrates were identified to genus if possible. UNK=unknown.











