# Recovery of Macroinvertebrate Communities after Remediation of Acid Mine Drainage



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*Recovery of Macroinvertebrate Communities after Remediation of Acid Mine Drainage*

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| Scott A Wissinger | 2 May 2011 | Kate Darby | 5.3.11 |
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Zachary L. Bell

**Pledge**

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## Abstract

Acid Mine Drainage (AMD) is a type of water pollution that affects many mining regions in Pennsylvania. The Slippery Rock Creek Watershed has installed 12 passive wetland treatment systems to treat mine drainage before it enters streams. Treatment wetlands have increased the water quality of the streams, but increase is not always accompanied by recovery of macroinvertebrates. The purpose of this study was to determine the degree to which macroinvertebrate communities in AMD remediated streams have recovered. I compared macroinvertebrates to pH and yellow boy of untreated, treated, and reference streams in Slippery Rock Creek. I hypothesized that reference streams would be characterized with more diversity of species and the presence of sensitive species. I also hypothesize that streams with high pH and no yellow boy will contain higher diversity and sensitive species than those with yellow boy. By sampling 12 sites in three different branches I found that reference streams were populated with more diversity and sensitive species. Streams with higher pH had more diversity and sensitive species. I also found that yellow boy negatively affected the macroinvertebrate community. Without the removal of yellow boy from the stream, reclamation and restoration may never reach its full potential.

# INTRODUCTION

*Problem: Acid Mine Drainage*

Acid Mine Drainage (AMD) is a long-term water pollutant that concerns many of the mining regions in the United States. Coal mines that drain water from abandoned or active coal mines contain chemicals that can pollute local streams. (O'Halloran et al.) These chemicals that come from coal water are sulfuric acid, acid salts, iron, aluminum, and other chemicals. When materials such as pyrite are exposed to oxygen and water these pollutants are formed. Pyrite ($FeS_2$) is contained in rock strata that are associated with coal seams. When mine wastes contains sulfides and are exposed to oxygen and water the sulfide minerals become oxidized by microbial and chemical processes. The results of these products contaminate surface waters and are responsible for acidification, heavy metal pollution, and sedimentation. (Earle and Callaghan, 2010) Low pH, high conductivity, sulfate concentration, and high heavy metals are characteristics of acid mine drainage.

AMD consists of two major factors. The two major factors are hydrogen ion acidity and mineral acidity which is related to manganese, iron, and aluminum. Ferric precipitate ("yellow boy") brings a major problem to stream ecosystems because it can destroy the habitat for organisms by covering the substrate in a thick yellow-orange color of precipitate. (Kanuckel, 2003) Ferric precipitate could possibly be the biggest pollutant of biological recovery due to its ability to destroy the habitat. Iron, pH, manganese, and aluminum usually account for the greater part of acidity with acid mine drainage from coal mines in western Pennsylvania. (Niyogi et al. 2001)

Acid Mine Drainage degrades more than 12,000 km of streams in the Appalachian Region of Northeast United States. The Appalachian Region that is mostly impacted is western Pennsylvania and West Virginia estimated at 80%. The estimated cost of AMD reclamation in Pennsylvania is over $15 billion of affected watersheds. There are many problems associated with mine drainage. (DeNicola & Stapleton, 2001) AMD can contaminate drinking water, disrupt growth and reproduction of aquatic plants and animals, and corrode infrastructure such as bridges due to acid.

*Acid Mine Drainage Solutions*

There are two types of ways to treat acid mine drainage. AMD can be treated passively or actively. Treating AMD passively uses bogs and marshes that are natural or constructed. Aerobic and anaerobic wetland processes help to successfully treat acid mine drainage. (Skousen et al. 1992) Aerobic wetland systems are designed to help the process of oxidation by using vegetation, shallow waters, and an abundance of surface flow. Anaerobic wetlands require that the acidic water flows through organic materials that are under anaerobic conditions. There are a variety of organic materials that can be used for anaerobic wetlands. All materials must contain sulfate reducing bacteria due to the chemicals. (Baker et al. 1990) The high levels of sulfate in the process have the ability to remove the metals within the wetlands.

An active treatment system requires constant monitoring and maintenance during the process. The traditional use of active treatment systems uses a process of aeration. (Flannagan et. al, 1993) The materials then settle and then are filtered out of the system. This process is not very effective in the removal of harmful chemicals. The addition of

chemicals during the process makes it effective, but the costs of building treatment plants, consumption of chemicals, and maintaining the system are high. (Mitsch and Wise, 1997) (Skouson et al. 1992) (Treacy and Timpson 1999)

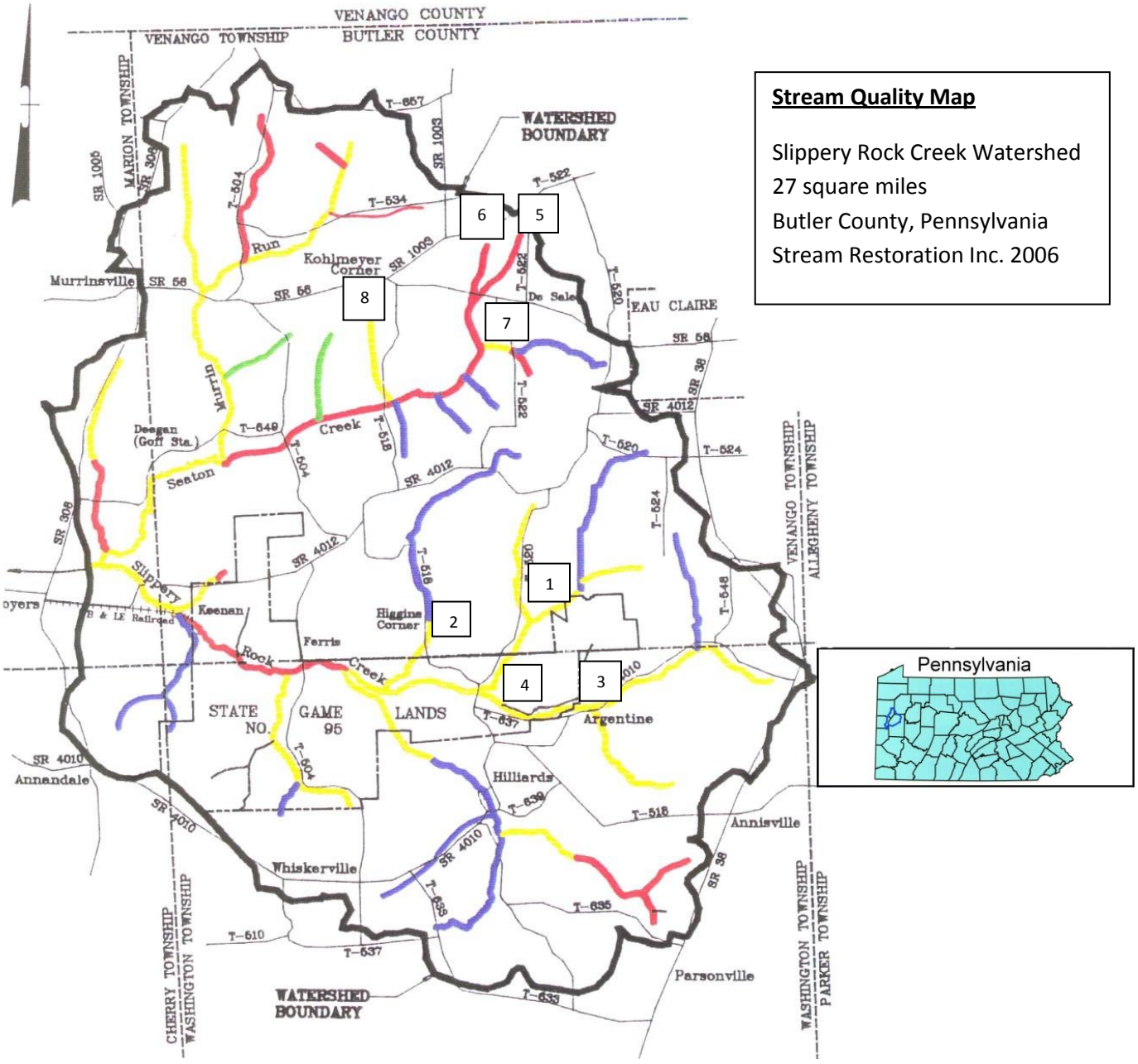
*Do the Solutions Completely Restore Streams?*

The treatment systems that are constructed are put in place to remediate streams that are impacted by acid mine drainage. These treatment systems have been put into operation, but there have been little to no work after to determine if these streams have been able to achieve normal levels. The treatment systems are used to restore streams to their original service and so diversity could return to the ecosystem. The few studies that have been used to explain the recovery of streams affected by AMD have not been able to evaluate the effectiveness of treatment systems. There is no standard criterion for how to quantify yellow boy. All the impacted streams have different response variables that have been assessed. The quality of the streams can be determined but for biomonitoring there is no component for AMD remediated streams. (DeNicola and Stapleton, 2001)

*Slippery Rock Creek*

The Slippery Rock Creek is a tributary of the upper Ohio River watershed of western Pennsylvania. The creek drains an area of approximately 1100 $km^2$. The Slippery Rock Creek watershed has been severely impacted by acid mine drainage from coal mining activities. (Figure 1A & 1B) Historically the headwater streams were impacted by coal mining activity in a 70 $km^2$ area. The pH values of the headwater streams have shown to range 3-6. The creek has high concentrations of iron (Fe) and low diversity of organisms. The land has since been reclaimed and has installed 12 passive treatment systems. The water

quality of the Slippery Rock Creek has improved greatly since 30 years ago. At certain time dissolved metals can exceed USA and Pennsylvania water quality standards. This usually only occurs during high flow periods. The substratum has since been covered with yellow boy and resulted in a low density of macroinvertebrates. (DeNicola and Stapleton, 2001.)




Blue- Good
Green-Drainage Sensitive
Yellow- Abandoned Mine Drainage Impacted
Red- Polluted
Black- Undesignated

Figure 1A: Slippery Rock Creek Study

**Table 1: Site Numbers**

| Name | Branch | Site Number |
|---|---|---|
| Pisor Run | Main | 1 |
| Higgins Hill Bridge | Main | 2 |
| Argentine Run | Main | 3 |
| State Game Lands | Main | 4 |
| Northeast Headwater | Seaton | 5 |
| Northwest Headwater | Seaton | 6 |
| Smith Run | Seaton | 7 |
| McJunkin Road | Seaton | 8 |
| Rental Run on SR8 | North | 9 |
| McMurray Run | North | 10 |
| Rocky Springs | North | 11 |
| Black's Creek | North | 12 |

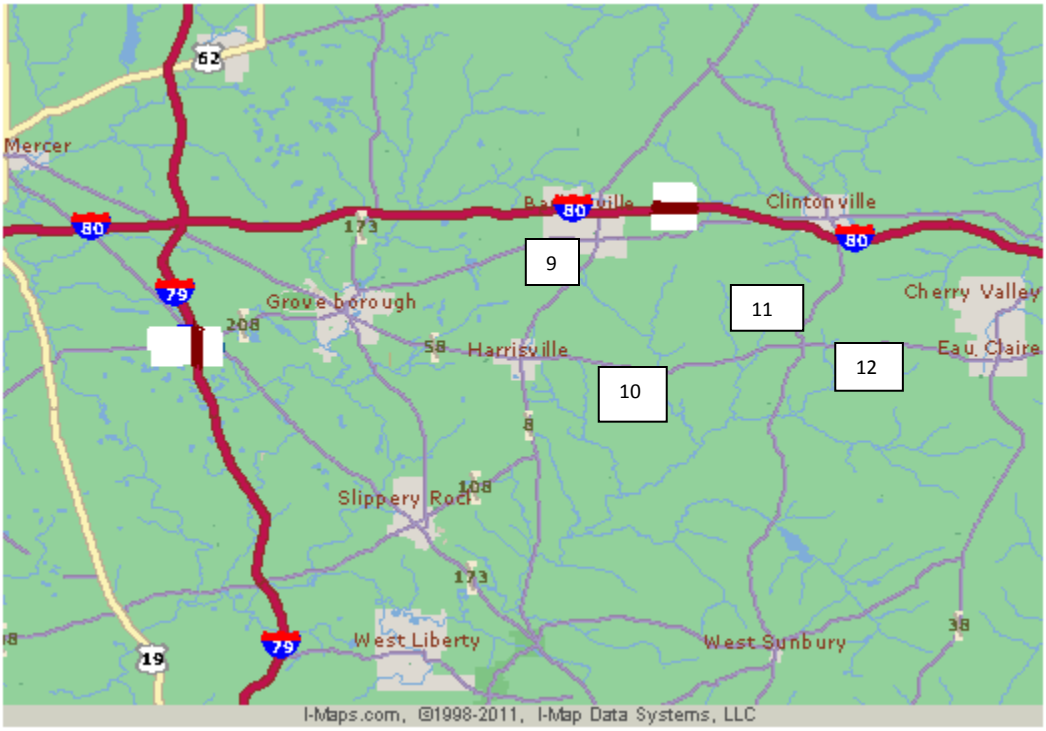



Figure 1B: I-MAPS, Harrisville, PA. 1998-2011

*Macroinvertebrate Biomonitoring*

In the past, stream conditions were determined by water chemistry. Stream conditions that show good water chemistry does not always describe the biology of the stream. (Karr and Chu 1999) The biological monitoring of benthic macroinvertebrate communities is a good source to determine the stream quality. Benthic macroinvertebrates is a good source because of their sensitivity to environmental stresses and also because of lack of mobility. Macroinvertebrates are a central position in the food chain and are important indicators of past and present stream conditions. They have the ability to describe water quality and biological quality. (Chadwick et al. 1986)

There are many ways to sample benthic macroinvertebrates and how to determine stream quality. Biotic indices have been developed to assess the impact of pollution in waterways. One way to determine stream quality is to use the Index of Biotic Integrity (IBI). This tool is used to quantify or grade stream integrity; it is most commonly used for biomonitoring. (Karr and Chu 1999) IBI's are composed of scales that grade number of species, number of tolerant species and total species diversity. The IBI's are separated into categories and produce a tangible score for stream integrity in terms of particular taxonomy response to stressors present in the stream. IBI's are scaled and adjusted for different streams, which provide stream ecologists and policy makers with flexible and recognizable results for assessing stream integrity. (Karr and Chu, 1999)

There are different variables that are commonly used to measure the recovery of biotic indices. Aquatic organisms respond in multiple ways to different types of pollutions. The responses depend on physiological characteristics of the organism and what type of

pollution whether it is chemical or physical. A stream that is free of pollution usually is diverse. Pollutants that occur decrease the diversity of streams. (Kanuckel, 2005) AMD affected streams decrease the diversity of organisms. Streams that are not affected by pollutants contain organisms that are tolerant and sensitive. AMD polluted streams show a low diversity of organisms and an absent of sensitive species.

There are many organisms that are tolerant of AMD which include alderfly, bloodworm midge, dytiscid beetles, caddisfly *Ptilostomis*, and megaloptera *Sialis*. Blackflies, crayfish, mayflies, stoneflies and species of caddisflies are moderately sensitive to high pollutants of acid mine drainage. Macroinvertebrates that are highly sensitive to acid mine drainage are Ephemeroptera, Plecoptera, and Odonata. These highly sensitive species have been seen to disappear from the ecosystem. (PA - DEP, 2009)

*Macroinvertebrate Recovery*

Streams that are remediated are difficult to determine the degree that they have recovered from a disturbance. The recovery rates of macroinvertebrates vary depending on species. The order of appearance in streams affected by AMD would be Trichoptera, Plecoptera, and Ephemeroptera. Chironomids is the one species that shows an increase in streams affected. (Winner et al. 1980)

*Purpose of Study*

The purpose of this study was to determine the degree to which AMD remediated streams have recovered. I compared macroinvertebrates to pH and yellow boy of untreated, treated, and reference streams in Slippery Rock Creek. I hypothesized that reference streams

will have more diversity of species and sensitive species. I also hypothesized that streams with high pH and no yellow boy will contain higher diversity and sensitive species.

## Methods

*Site Selection*

Slippery Rock Creek a tributary to the Ohio River watershed is the selected site location. It is severely impacted by Acid Mine Drainage predominantly from coal mining activity. (DeNicola & Stapleton, 2001) There were 12 sits selected for this study; four from North Branch (Rental Run on SR 8, McMurray Run, Rocky Springs, and Black's Creek at SR 58), four from Seaton Branch (Smith Run, Northwest HW, Northeast HW, and McJunkin Road), and four from Main Branch (Argentine Headwater, State Game Land Gate, Higgins Hill Bridge, and Pizer Run). All of the sites in this study contained at least one riffle/run system. The reference sites are all found in North Branch (Figure 1B); while treated sites are found in Seaton Branch (all sites) and Main Branch (SGL Gate, and Higgins Hill Bridge). Untreated sites are found in Main Branch (Argentine Headwater and Pizer Run). (Figure 1A) (Table 1)

*Biomonitoring*

Macroinvertebrate samples were taken at each stream site. They were sampled by using a 500 micron D-frame kick-net downstream and kicking one square meter for one minute upstream. Six samples were taken at each site from a riffle sequence. Once all macro-invertebrates were collected the contents of the net were preserved in 95% ethanol and analyzed in the laboratory using the Pennsylvania Department of Environmental Protection BIBI protocol. (Goetz, 2009) All macroinvertebrates were collected between November 10$^{th}$ and December 18$^{th}$.

*Water Chemistry*

General water chemistry was taken at each site. Measurements of pH were taken at each site in the field with a handheld Oakton Meter.

*Land Use*

The discharge was measured using a Marsh-Mcbirney Model 2000 Portable Flowmeter.

*Statistical Analysis*

The data were analyzed with StatView. The statistics of the measurements were made into one-way ANOVA graphs, Fisher's PLSD graphs, and a regression plot graph. All field and lab samples give explanation, how each has an effect on the other.

## Results

*Physical Variation between Sites*

When analyzing the sites within the study area I found significant differences. I found a difference in pH values, taxa richness, EPT richness, % sensitive, and BIBI scores. The pH differed among the sites (Fisher's PLSD <0.05 $F_{2,9} = 8.2$ P = 0.09). Sites containing No AMD differed from Remediated (P = 0.004) and Untreated (P = 0.01). Remediated compared to Untreated there was no significant difference. (P = 0.9) (Figure 2)

The taxa richness differed between the sites ($F_{2,9} = 19.1$; P = 0.0006). Sites with No AMD had a significantly higher taxa richness from (Fisher's PLSD < 0.05) Remediated (P = 0.0002) and Untreated (P = 0.002). Remediated compared to Untreated did not significantly differ. (P = 0.7) (Figure 3)

The EPT richness differed between the sites ($F_{2,9} = 8.4$; P = 0.0086). Sites with No AMD had a significantly higher EPT richness from (Fisher's PLSD < 0.05) Remediated (P = 0.004) and Untreated (P = 0.01). Remediated compared to Untreated did not significantly differ. (P = 0.6) (Figure 4)

The % Sensitive species differed between the sites ($F_{2,9} = 4.1$; P = 0.05). Sites with No AMD had a significantly higher % sensitive from (Fisher's PLSD < 0.05) Untreated (P = 0.02) sites. Sites with No AMD did not have a significant difference compared to Remediated (P = 0.08). Remediated also did not have a significant difference compared to Untreated. (P = 0.18) (Figure 5)

The BIBI Scores differed between the sites ($F_{2,9}$ =8.8; P = 0.007). Sites with No AMD had a significant higher BIBI score (Fisher's PLSD < 0.05) than Remediated (P = 0.008) and Untreated sites (P = 0.004). Remediated compared to Untreated did not significantly differ. (P = 0.2) (Figure 6)

*Physical Variation between Branches*

When analyzing each branch within the study area I found differences in many areas. I found a difference in pH values, taxa richness, EPT richness, % sensitive, and BIBI scores. The pH differed among the three branches (one-way ANOVA main effect of Branch $F_{2,9}$ = 9.3; P = 0.006), with North Branch having a significantly higher pH from the other two branches. (Figure 7)

The taxa richness differed between the three branches ($F_{2,9}$ = 20.6; P = 0.004). North had a significantly higher taxa richness from (Fisher's PLSD < 0.05) Main Branch (P = 0.0007) and Seaton Branch (P = 0.0002). Main Branch compared to Seaton Branch did not differ in taxa richness (P = 0.41). (Figure 8)

The EPT richness differed between the three branches ($F_{2,9}$ = 8.4; P = 0.0086). North had a significantly higher EPT richness from (Fisher's PLSD < 0.05) Main Branch (P = 0.009) and Seaton Branch (P = 0.004). Main Branch compared to Seaton Branch did not differ in EPT Richness significantly (P = 0.6). (Figure 9)

The % sensitive species between the three branches showed a significant difference (one-way ANOVA main effect of Branch $F_{2,9}$ = 5.7; P = 0.2). North Branch shows a

significant difference between both branches. Seaton Branch compared to Main Branch shows a significant difference. (Figure 10)

The BIBI scores between the three branches showed a significant difference ( one-way ANOVA main effect of Branch $F_{2,9} = 7.6$; $P = 0.01$). North Branch shows a significant difference between both branches. Seaton Branch compared to Main Branch shows a difference but not that significant. (Figure 11)

*Effects of pH and Yellow Boy on Benthic Invertebrate Communities*

Analyzing the effect of pH and Yellow Boy on the benthic invertebrate communities it showed a significant regression between EPT richness and pH. The taxa richness compared to streams with Yellow Boy and without showed a significant difference.

The regression of EPT richness vs. pH shows a significance (linear regression plot EPT richness vs. pH $F_{1,10} = 10.1$; $P = 0.09$). The regression shows a higher amount of EPT richness as the pH rises in the stream ($Y = -16.461 + 2.935*X$; $R^2 = .503$). (Figure 12)

The regression of BIBI vs. pH shows a significance (linear regression plot BIBI vs. pH $F_{1,10} = 7.3$; $P = 0.02$). The regression shows a higher amount of BIBI scores as the pH rises in the stream ($Y = -55.814 + 12.883 * X$; $R^2 = .424$). (Figure 13)

The regression of Total Richness vs. pH shows a significance (linear regression plot Total Richness vs. pH $F_{1,10} = 7.3$; $P = 0.02$). The regression shows a higher amount of Total Richness as the pH rises in the stream ($Y = -31.93 + 5.754 * X$; $R^2 = .592$). (Figure 13)

The taxa richness between streams with yellow boy and no yellow boy shows a significant difference (Unpaired t-test for Richness P = 0.009). The taxa richness is significantly greater in streams without yellow boy than those with. (Figure 8)

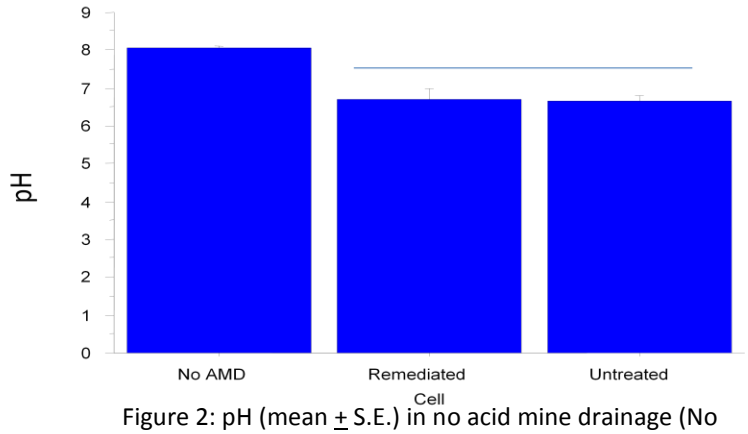



Figure 2: pH (mean ± S.E.) in no acid mine drainage (No AMD), remediated, and untreated tributaries of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

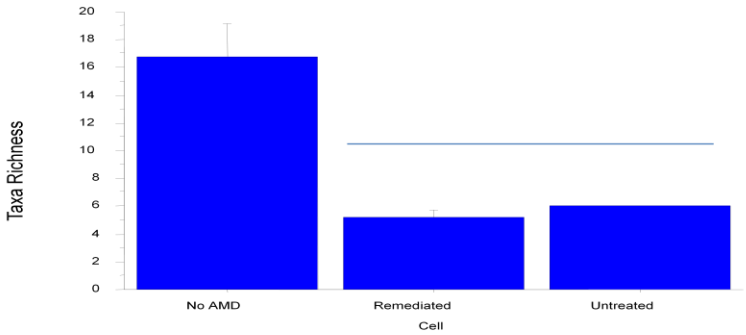



Figure 3: Taxa Richness (mean ± S.E.) in no acid mine drainage (No AMD), remediated, and untreated tributaries of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

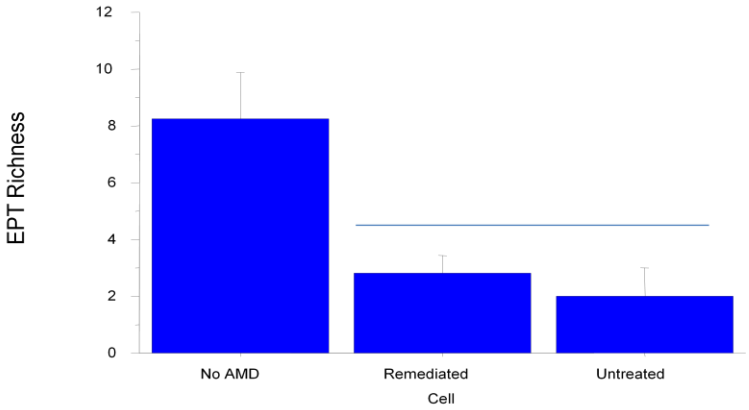



Figure 4: EPT Richness (mean ± S.E.) in no acid mine drainage (No AMD), remediated, and untreated tributaries of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

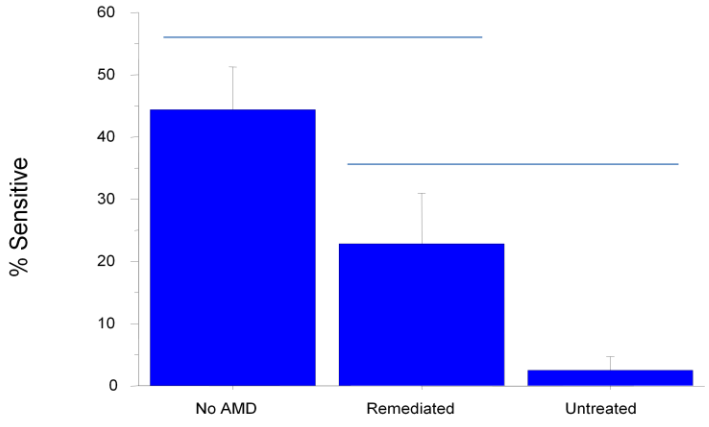



Figure 5: % Sensitive (mean ± S.E.) in no acid mine drainage (No AMD), remediated, and untreated tributaries of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

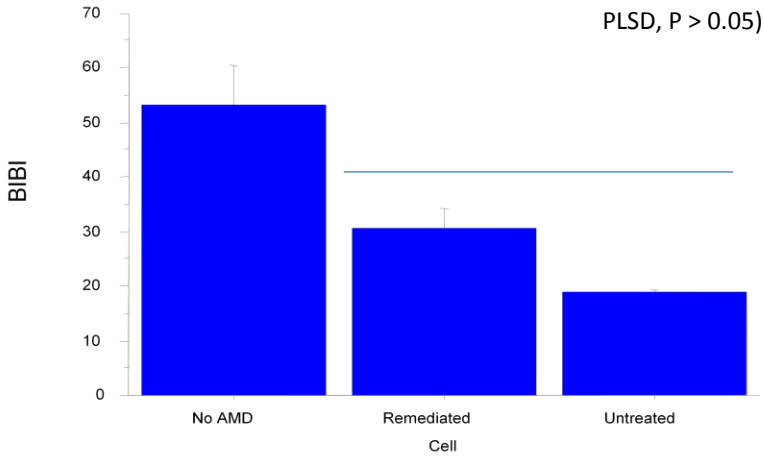



Figure 6: pH (mean ± S.E.) in no acid mine drainage (No AMD), remediated, and untreated tributaries of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

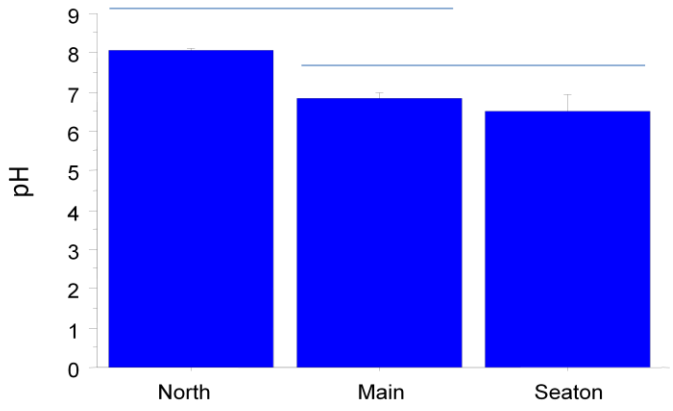



Figure 7: pH (mean ± S.E.) in North, Main, and Seaton Branches of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

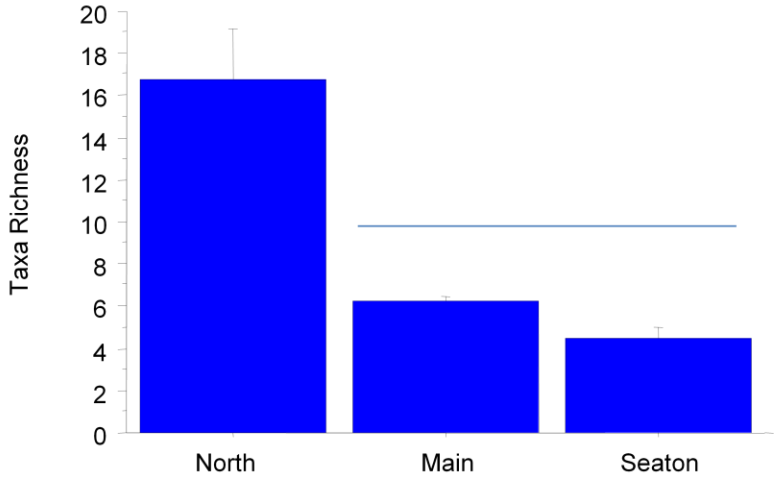



Figure 8: Taxa Richness (mean ± S.E.) in North, Main, and Seaton Branches of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

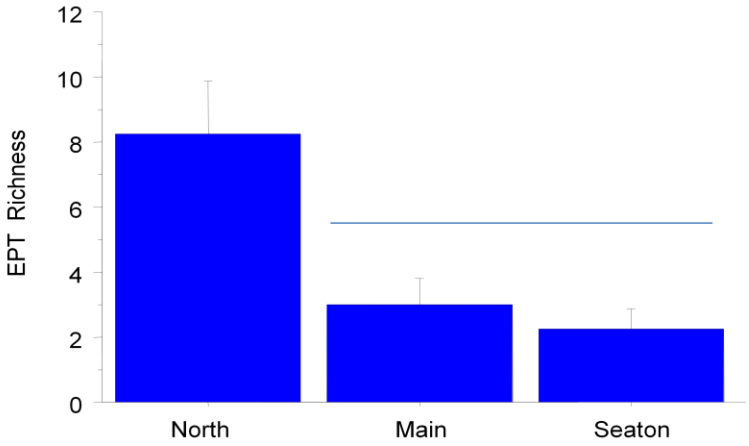



Figure 9: EPT Richness (mean ± S.E.) in North, Main, and Seaton Branches of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

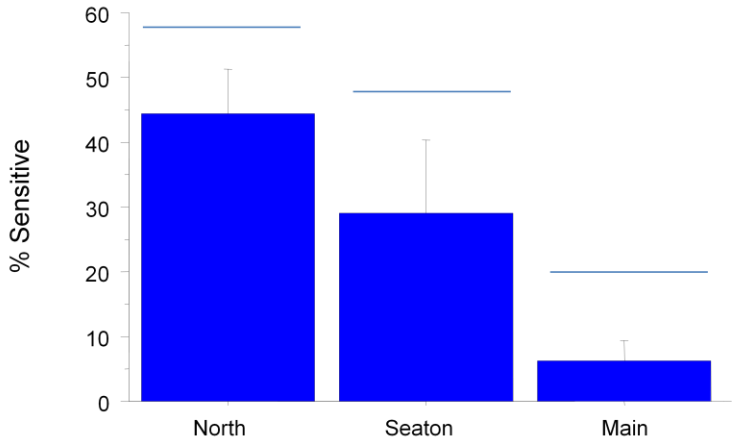



Figure 10: % Sensitive (mean ± S.E.) in North, Main, and Seaton Branches of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

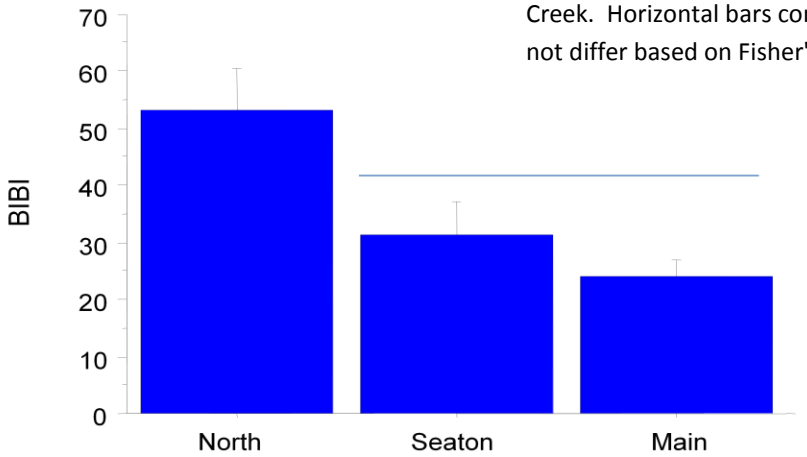



Figure 11: BIBI (mean ± S.E.) in North, Main, and Seaton Branches of Slippery Rock Creek. Horizontal bars connect means that do not differ based on Fisher's PLSD, P > 0.05).

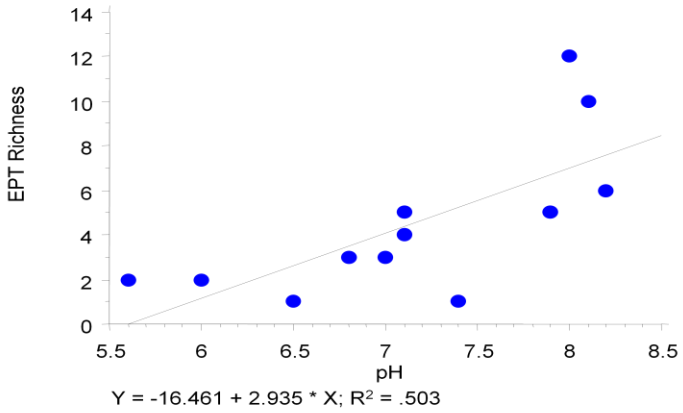



Figure 12: Linear Regression of pH vs. EPT Richness P=0.009

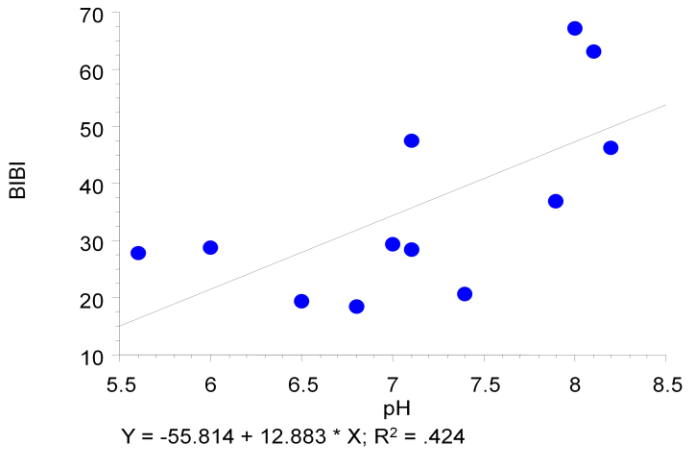



Figure 13: Linear Regression of pH vs. BIBI P=0.02

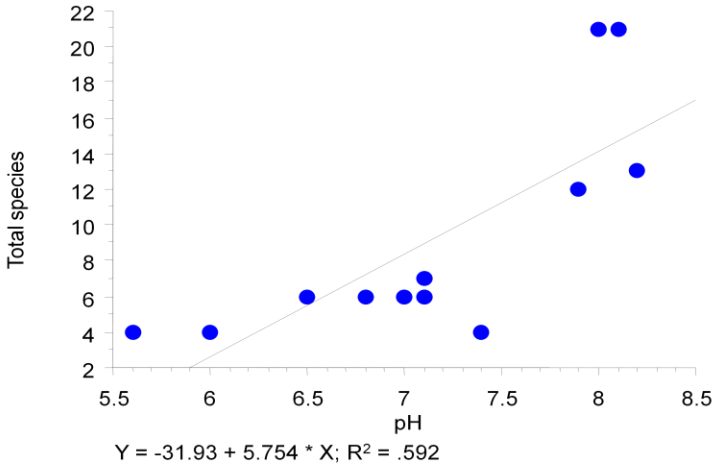



Figure 14: Linear Regression of pH vs. Total Species P=0.003

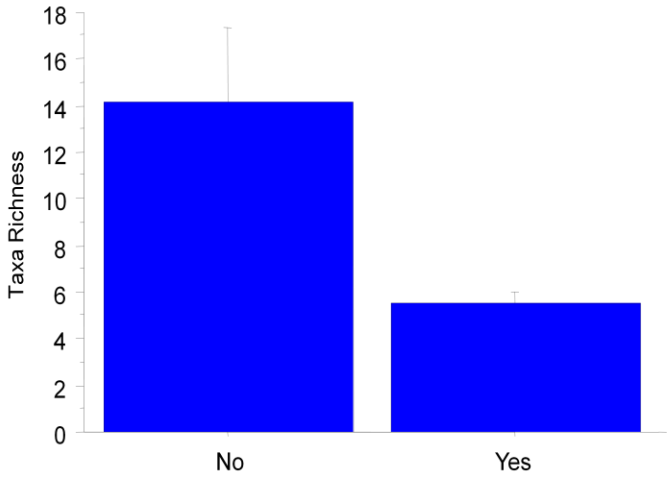



Figure 15: Taxa Richness vs. Yellow Boy One-Way ANOVA main effect Yellow Boy

## Discussion

There were significant differences between the sites and branches of Slippery Rock Creek. The difference in pH levels shows that lower pH had a decrease in benthic macroinvertebrates. (DeNicola and Stapleton, 2001) The control branch (No AMD) (North Branch) was characterized with more diversity than the remediated sites and untreated sites. Though the water quality has improved from remediation it has not returned to that of the control branch.

The pH also showed a significant difference in EPT richness. EPT richness was significantly decreased downstream from the No AMD sites. (Appendix B for EPT data) A study done in Silver Bow Creek, Montana, has shown that after remediation it took 10 years for macroinvertebrates to repopulate the stream. This included Ephemeroptera, Plecoptera, and Trichoptera only after remediation had taken years of effect. (Chadwick et al. 1986) The pH levels of Seaton Branch and Main Branch have not reached the same level of North Branch.

EPT richness at the control branch (No AMD) was higher than the other two branches. Since these variables are good indicators of recovery, there does not appear to be a large amount of recovery at Slippery Rock Creek. The % sensitive species differed among the branches were the remediated sites had more than the untreated sites. The control branch ranked highest among the three. A study has shown that sensitive species are eliminated or reduced. (Kunuckel, 2005) Sensitive species did recover at the remediated sites showing signs of improvement.

The BIBI scores calculated (Appendix A for BIBI calculations) showed that the control branch scored higher than the remediated and untreated sites. (Figures 6 & 11) There was not a great score difference between the remediated sites and untreated sites. There was a significant difference between the control branch and the other branches.

Yellow Boy that was found at Seaton Branch and Main Branch showed to decrease taxa richness at the sites. DeNicola and Stapleton (2001) have found that taxa richness was not significantly different between polluted sites and control sites. I have found that taxa richness was significantly different from control and polluted sites. The control site yielded at least twice the amount of taxa then the remediated and untreated streams. Trichoptera found at sites containing yellow boy were covered with iron precipitate. A study that was done has shown that this has also occurred and covered the thorax and gills of macroinvertebrates. (Vuori, 1995)

After all the data were compiled, I concluded that there were seven significant results. I have found that Seaton Branch has more sensitive species but less overall taxa richness than that of Main Branch. pH levels at Seaton Branch were lower than Main Branch but had more sensitive species. I suspect that this is due to yellow boy that has covered the substratum and disturbing the ecosystem. DeNicola and Stapleton (2001) have found that though water quality has increased an abundance of iron precipitate can disturb the ecosystem. The recolonization of macroinvertebrates may always be lower or never reach full growth until the substratum of the streams have been remediated and iron precipitate reduced. From this study I have concluded that until yellow boy has been removed completely from the streams, recovery might not reach full potential.

Future studies could use the data from this study to see if there has been more improvement to the reaches of the Slippery Rock Creek tributaries that have had remediation. If the benthic macroinvertebrate community was monitored on a yearly basis then different variables could be found and a more detailed list of what sensitive species are recovering or tolerant species have populated. I would also recommend that samples of iron precipitate be taken to see the degree that yellow boy has covered the substratum. This would be useful to determine if the yellow boy is the main cause for depletion of macroinvertebrates, from what DeNicola and Stapleton (2001) have found.

Changes that I would make to this study would be to find better riffle/run sequences. Sedimentation was high at many parts of the stream. Some sites had trash that was thrown into the streams. I found garbage bags in some of the sites while collecting samples. New sites might be found that are not full of sediment.

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Appendix A:
PA- DEP RBP Metrics Table and Support Materials Index of Biological Integrity for Wadeable, Freestone, Riffle Run Streams in Pennsylvania

**Introduction**

The Pennsylvania Department of Environmental Protection (DEP) developed an index of biotic integrity (IBI) for benthic macroinvertebrate communities in Pennsylvania's wadeable, freestone, riffle-run type streams as a scientifically credible biological assessment tool. This indicator assists in guiding and evaluating legislation, policy and management strategies as well as setting goals for aquatic resources by enabling direct quantification of important ecological attributes along a gradient of biological conditions and ecosystem stressors (Davis and Simon 1995; Davies and Jackson 2006; Hawkins 2006). This indicator serves as a measure of the extent to which anthropogenic stressors impair the capability of a stream to support a healthy aquatic community (Davis and Simon 1995).

**Biological Sampling Methods**

This IBI applies to benthic macroinvertebrate samples collected any time of the year from wadeable, freestone, riffle-run streams in Pennsylvania using a D-frame net with 500-micron mesh. Field sampling and laboratory methods are more fully described in the Department's Standardized Biological Field Collection and Laboratory Methods, Section V (Pa DEP 2003). Sampling biologists' composite six kicks from riffle areas distributed throughout a 100-meter stream reach, working progressively upstream, with each kick disturbing approximately one square meter immediately upstream of the net for approximately one minute to an approximate depth of 10 cm, as substrate allows. Composited samples are preserved with 95% ethanol in the field and transported back to the laboratory for processing. In the lab, each composited sample was placed into a 3.5 inch deep rectangular pan (measuring 14" long x 8" wide on the bottom of the pan) marked off into 28 foursquare
inch (2" x 2") grids. Four of the grids are randomly selected, their contents are extracted using a four-square inch circular "cookie cutter," and placed into another identical empty pan. From this second pan, organisms are picked from randomly selected grids until a 200-organism sub-sample (+/- 40 organisms) is obtained. If less than 160 identifiable organisms are sub-sampled from the original four grids, additional grids are extracted from the first pan, transferred to the second pan and picked until the target number of organisms are obtained. If more than 240 identifiable organisms are sub-sampled from the original four grids, one grid at a time is randomly selected and removed from the second pan until the target number of organisms is obtained. Any grids selected during this entire process are picked in their entirety and the total numbers of grids selected for each part of the sub-sampling process are recorded.

Organisms in the sub-sample are identified and counted. Midges are identified to the family level of Chironomidae. Snails, clams and mussels are all also identified to family levels. Roundworms and proboscis worms are identified to the phylum levels of Nematoda and Nemertea, respectively. Moss animacules are identified to the phylum level of Bryozoa. Flatworms and leeches are identified to the class levels of Turbellaria and Hirudenia, respectively. Segmented worms, aquatic earthworms, and tubificids are identified to the class level of Oligochaeta. All water mites are identified as Hydracarina, an artificial taxonomic

grouping of several mite superfamilies. All other macroinvertebrates are identified to genus level. Most of the samples used to develop the IBI were taken from relatively small, mostly first through third order riffle-run type streams draining less than 25 square miles, so this IBI should be applied with discretion to other stream types (e.g., limestone type streams) and larger stream/river systems. Currently, DEP does not apply any regionally-based classification to wadeable, freestone, riffle-run streams in the Commonwealth for purposes of applying this IBI.

**The Metrics**

A number of different metric combinations were evaluated during index development and the following six metrics were selected for inclusion as core metrics in the IBI based on various performance characteristics. These six metrics all exhibited a strong ability to distinguish between reference and stressed conditions. In addition, these six metrics measure different aspects of the biological communities represented by the sub-samples, and when used together in a multimetric index, they provide a solid foundation for assessing the biological condition of benthic macroinvertebrate assemblages in Pennsylvania's wadeable freestone riffle-run stream ecosystems.

**Total Taxa Richness**

This taxonomic richness metric is a count of the total number of taxa in a sub-sample. Generally, this metric is expected to decrease with increasing anthropogenic stress to a stream ecosystem, reflecting loss of taxa and increasing dominance of a few pollution tolerant taxa. Other benefits of including this metric include its common use in many biological monitoring and assessment programs in other parts of the world as well as its ease of explanation and calculation.

**Ephemeroptera + Plecoptera + Trichoptera Taxa Richness (Pollution Tolerance Value 0 - 4 only)**

This taxonomic richness metric is a count of the number of taxa belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) in a sub-sample – common names for these orders are mayflies, stoneflies, and caddisflies, respectively. The aquatic life stages of these three insect orders are generally considered sensitive to, or intolerant of, pollution (Lenat and Penrose 1996); in fact, this metric only counts EPT taxa with pollution tolerance values (PTVs) of 0 to 4, excluding a few of the most tolerant mayfly and caddisfly taxa. This metric is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting the loss of taxa from these largely pollution-sensitive orders. This metric has a history of use across the world and is relatively easy to use, explain and calculate (Lenat and Penrose 1996).

**Beck's Index, version 3**

This taxonomic richness and tolerance metric is a weighted count of taxa with PTVs of 0, 1, or 2. The name and conceptual basis of this metric are derived from the water quality work of William H. Beck in Florida (Beck 1955). This metric is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting the loss of pollution-sensitive taxa. It should be noted that the version of the Beck's Index metric used for this project, although similar in name and concept,

differs slightly in its calculation from the Beck's Index used in DEP's multihabitat protocol for assessing biological condition of low gradient pool-glide type streams.

**Shannon Diversity**
This community composition metric measures taxonomic richness and evenness of individuals across taxa of a sub-sample. This metric is expected to decrease in values with increasing anthropogenic stress to a stream ecosystem, reflecting loss of pollution-sensitive taxa and increasing dominance of a few pollution-tolerant taxa. The name and conceptual basis for this metric are derived from the information theory work of Claude Elwood Shannon (Shannon 1968).

**Hilsenhoff Biotic Index**

This community composition and tolerance metric is calculated as an average of the number of individuals in a sub-sample, weighted by PTVs. Developed by William Hilsenhoff, the Hilsenhoff Biotic Index (Hilsenhoff 1977, 1987, 1988; Klemm et al. 1990) generally increases with increasing ecosystem stress, reflecting increasing dominance of pollution-tolerant organisms.

**Percent Sensitive Individuals (PTV 0 – 3)**

This community composition and tolerance metric is the percentage of individuals with PTVs of 0 to 3 in a sub-sample and is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting loss of pollution-sensitive organisms. To find PTV values look in PA-DEP Division of Water Quality Assessment and Standards. (A Benthic Index of Biotic Integrity for Wadeable Freestone Riffle-Run Streams in Pennsylvania)

| Metric Name | Standardization Value | *Observed Value* | Standardized Metric Score | Adjusted Metric Score |
|---|---|---|---|---|
| Total Taxa Richness | 33 | *A* | A/33 | |
| EPT Taxa Richness (PTV 0 - 4 only) | 19 | *B* | B/19 | |
| Beck's Index (version 3) | 38 | *C* | C/38 | |
| Shannon Diversity Index | 2.86 | *D* | D/2.86 | |
| Hilsenhoff Biotic Index | 1.89 | *E* | (10-E)/(10-1.89) | |
| Percent Intolerant Individuals (PTV 0 - 3) | 84.5 | *F* | F/84.5 | |
| | | **2008 FREESTONE 6D IBI Score =** | | |

For larger wadeable freestone riffle-run type streams, DEP believes more samples are necessary to accurately establish ALU attainment and impairment benchmarks. Given the nature of flowing water bodies as gradually changing continuums, it is difficult to define a specific numeric cutoff to separate larger streams from smaller streams. However, the present

dataset suggest that scores for some index metrics begin to decline for reference-quality streams drainage areas that reach the 25 to 50 square mile range. Workshops conducted by DEP confirm that biological expectations or potential for most of the relatively pristine larger freestone streams in Pennsylvania are less than the biological expectations or potential for the relatively pristine smaller freestone streams. The use assessment decision process and accompanying attainment/impairment benchmarks set forth above are intended as general guidelines, not as hard-and-fast rules. While the above guidelines will provide an accurate assessment of benthic macroinvertebrate community condition for the vast majority of samples collected from wadeable, freestone, riffle-run streams in Pennsylvania, there will be instances where a biologist's local knowledge of conditions may warrant a decision not arrived at using these guidelines. For instance, if a sample is heavily dominated by Simuliidae or Chironomidae larvae, often times this will make the metric and IBI scores difficult to interpret and the investigating biologist must rely on a more qualitative analysis of the metric scores and sample composition to arrive at an assessment decision. Similarly, samples from streams in areas receiving a substantial amount of flow from groundwater attributable to limestone geology are naturally expected to have less diversity than "true freestone" streams, so use attainment benchmarks may be justifiably relaxed for samples from these types of streams.

**References for PA- DEP RBP Metrics Table and Support Materials Index of Biological Integrity for Wadeable, Freestone, Riffle Run Streams in Pennsylvania**

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Appendix B: Raw Data

| Name | Branch | Total Taxa Richness | EPT Taxa Richness | Beck's Index |
|---|---|---|---|---|
| Pisor Run | Main | 6 | 3 | 3 |
| Higgins Hill Bridge | Main | 7 | 5 | 3 |
| Argentine Run | Main | 6 | 1 | 0 |
| State Game Lands | Main | 6 | 3 | 6 |
| Northeast Headwater | Seaton | 4 | 2 | 0 |
| Northwest Headwater | Seaton | 4 | 2 | 0 |
| Smith Run | Seaton | 4 | 1 | 0 |
| McJunkin Road | Seaton | 6 | 4 | 8 |
| Rental Run on SR8 | North | 13 | 6 | 8 |
| McMurray Run | North | 12 | 5 | 2 |
| Rocky Springs | North | 21 | 10 | 7 |
| Black's Creek | North | 21 | 12 | 16 |

| Name | Branch | Shannon Diversity Index | Hilsenhoff Index | % Sensitive Species | BIBI |
|---|---|---|---|---|---|
| Pisor Run | Main | 0.299 | 5.81 | 4.76 | 18.3 |
| Higgins Hill Bridge | Main | 1.5117 | 5.35 | 4.54 | 28.5 |
| Argentine Run | Main | 1.6675 | 7.25 | 0 | 19.3 |
| State Game Lands | Main | 1.1702 | 4.51 | 15.5 | 29.5 |
| Northeast Headwater | Seaton | 1.2799 | 3.8 | 20 | 27.9 |
| Northwest Headwater | Seaton | 1.5101 | 3.8 | 18.2 | 28.9 |
| Smith Run | Seaton | 0.9639 | 5.5 | 14.8 | 20.7 |
| McJunkin Road | Seaton | 1.4525 | 1.06 | 63 | 47.6 |
| Rental Run on SR8 | North | 2.0074 | 3.93 | 34 | 46.2 |
| McMurray Run | North | 1.5044 | 4.67 | 30 | 37 |
| Rocky Springs | North | 2.4834 | 2.65 | 56 | 63.1 |
| Black's Creek | North | 2.5021 | 3.53 | 57 | 67.3 |

Appendix B continued: Raw Data

| Name | Branch | Discharge | pH | Yellow Boy |
|---|---|---|---|---|
| Pisor Run | Main | 0.4 | 6.8 | Yes |
| Higgins Hill Bridge | Main | 1.52 | 7.1 | Yes |
| Argentine Run | Main | 0.55 | 6.5 | Yes |
| State Game Lands | Main | 0.89 | 7 | Yes |
| Northeast Headwater | Seaton | 0.25 | 5.6 | Yes |
| Northwest Headwater | Seaton | 0.1 | 6 | Yes |
| Smith Run | Seaton | 0.11 | 7.4 | No |
| McJunkin Road | Seaton | 0.04 | 7.1 | Yes |
| Rental Run on SR8 | North | 0.48 | 8.2 | No |
| McMurray Run | North | 0.29 | 7.9 | No |
| Rocky Springs | North | 0.19 | 8.1 | No |
| Black's Creek | North | 0.52 | 8 | No |

Appendix B (Continued): Raw Data

| | North Branch | | Seaton Branch | | Main Branch |
|---|---|---|---|---|---|
| **Ephemeroptera** | **86** | | **0** | | **0** |
| **Heptageniidae** | **60** | | **0** | | **0** |
| *Macdannoia* | 46 | | 0 | | 0 |
| *Stenacron* | 6 | | 0 | | 0 |
| *Stenonema* | 8 | | 0 | | 0 |
| **Metretopodidne** | **3** | | **0** | | **0** |
| *Siphloplecton* | 3 | | 0 | | 0 |
| **Ephemeridae** | **23** | | **0** | | **0** |
| *Ephermera* | 23 | | 0 | | 0 |

| | North Branch | | Seaton Branch | | Main Branch |
|---|---|---|---|---|---|
| | | | | | |
| **Plecoptera** | **93** | | **121** | | **6** |
| **Taeniopterygidae** | **16** | | **0** | | **0** |
| *Taeniopteryx* | 16 | | 0 | | 0 |
| **Pteronarcyidae** | **1** | | **0** | | **0** |
| *Pteronarcys* | 1 | | 0 | | 0 |
| **Chloroperlidae** | **6** | | **0** | | **6** |
| *Utaperla* | 6 | | 0 | | 6 |
| **Perlodidae** | **10** | | **0** | | **0** |
| *Remus* | 6 | | 0 | | 0 |
| Isoperla | 1 | | 0 | | 0 |
| *Hansonoperla* | 3 | | 0 | | 0 |
| **Capniidae** | **4** | | **34** | | **0** |
| *Allocapnia* | 0 | | 10 | | 0 |
| *Capnia* | 4 | | 24 | | 0 |
| **Leuctridae** | **56** | | **87** | | **0** |
| *Zealeuctra* | 0 | | 38 | | 0 |
| *Leuctra* | 56 | | 49 | | 0 |

Appendix B (continued): Raw Data

| | **North Branch** | | **Seaton Branch** | | **Main Branch** |
|---|---|---|---|---|---|
| | | | | | |
| **Trichoptera** | **293** | | **18** | | **126** |
| **Philopotamidae** | **81** | | **0** | | **6** |
| *Lype* | 18 | | 0 | | 0 |
| *Chimarra* | 1 | | 0 | | 0 |
| *Dolophilodes* | 29 | | 0 | | 0 |
| *Wormaldia* | 33 | | 0 | | 6 |
| **Hydroptilidae** | **16** | | **13** | | **0** |
| *Ithytrichia* | 2 | | 13 | | 0 |
| *Polycentropodidae* | 2 | | 0 | | 0 |
| *Hydroptila* | 1 | | 0 | | 0 |
| *Orthotricia* | 11 | | 0 | | 0 |
| **Hydropsychidae** | **175** | | **5** | | **120** |
| *Patamyia* | 56 | | 3 | | 17 |
| *Hydropsyche* | 0 | | 0 | | 30 |
| *Macrostemum* | 59 | | 2 | | 2 |
| *Cheumatopsyche* | 60 | | 0 | | 81 |
| **Odontoceridae** | **16** | | **0** | | **0** |
| *Marilia* | 16 | | 0 | | 0 |
| **Phryganeidae** | **4** | | **0** | | **0** |
| *Oligostomis* | 4 | | 0 | | 0 |
| **Capniidae** | **1** | | **0** | | **0** |
| *Allocapnia* | 1 | | 0 | | 0 |