2004

Local and regional impairment of fish assemblages in a mined Appalachian watershed

Jason Gregory Freund
West Virginia University

Follow this and additional works at: <https://researchrepository.wvu.edu/etd>

Recommended Citation

Freund, Jason Gregory, "Local and regional impairment of fish assemblages in a mined Appalachian watershed" (2004). *Graduate Theses, Dissertations, and Problem Reports*. 2111.
<https://researchrepository.wvu.edu/etd/2111>

This Dissertation is protected by copyright and/or related rights. It has been brought to you by the The Research Repository @ WVU with permission from the rights-holder(s). You are free to use this Dissertation in any way that is permitted by the copyright and related rights legislation that applies to your use. For other uses you must obtain permission from the rights-holder(s) directly, unless additional rights are indicated by a Creative Commons license in the record and/ or on the work itself. This Dissertation has been accepted for inclusion in WVU Graduate Theses, Dissertations, and Problem Reports collection by an authorized administrator of The Research Repository @ WVU. For more information, please contact researchrepository@mail.wvu.edu.

Local and Regional Impairment of Fish Assemblages in a Mined Appalachian Watershed

Jason Gregory Freund

Dissertation submitted to the
Davis College of Agriculture, Forestry, and Consumer Science
at West Virginia University
in partial fulfillment of the requirements
for the degree of

Doctor of Philosophy
In
Forestry

Approved by

Dr. J. Todd Petty, Committee Chairperson

Dr. Kyle J. Hartman

Dr. Patricia M. Mazik

Dr. Michael P. Strager

Dr. Stuart Welsh

Division of Forestry

Morgantown, West Virginia
2004

Keywords: acid mine drainage (AMD), Cheat River (West Virginia), core/periphery view, fish assemblages, Index of Biotic Integrity (IBI), local and regional factors, multivariate models, predictive models, stream fish life history, watershed impairment

Copyright 2004 Jason G. Freund

Abstract

Local and Regional Impairment of Fish Assemblages in a Mined Appalachian Watershed

By Jason G. Freund

I used the Cheat River watershed, West Virginia, as a model system to examine the regional impacts of extensive localized stream impairment. Acid mine drainage (AMD) in the Cheat River watershed provided a unique opportunity to test the hypothesis that regional biological condition can be impacted through multiple local sources of degradation. Acid mine drainage is the result of the interaction among pyrite-rich coal, oxygen, and water. This interaction produces a solution that is generally low in pH and buffering capacity and high in heavy metals and sulfate and is toxic to aquatic life. Acid mine drainage also has the potential to isolate upstream reaches by acting as a barrier to dispersal and by reducing the source of colonizing individuals through the interaction between species life history and the toxic effects of AMD. Although the impacts of AMD are unique, other local stressors such as channelization, urbanization, and impoundments have the potential to produce similar regional impacts.

In order to test the local and regional impacts of mining-related discharge in the lower Cheat River basin, I examined fish and macroinvertebrate assemblage response to the direct and indirect effects of AMD. Fish assemblages were sampled in 66 unique stream segments within Cheat River tributaries of first through third Strahler order. Macroinvertebrate data were obtained from the West Virginia Division of Environmental Protection (WVDEP). Fish index of biotic integrity (IBI) and macroinvertebrate stream condition index (SCI) were used to quantify fish and macroinvertebrate assemblage response to impairment. Species richness and presence/absence models were developed in the non-AMD impacted upper basin and were applied to the lower basin. Additionally, I developed a core/periphery model to describe range of stream basin areas individual stream fish species. Fishes were classified by the size of streams their core contained (i.e. small, intermediate, large, or no core) and the extent of their core (i.e., narrow or wide). Results from these quantitative methods allowed me to test the regional effects of AMD in the lower Cheat River basin.

As a result of the alteration of species richness patterns and notable deviations from core/periphery and presence/absence model expectations, I concluded that the lower Cheat River watershed is impaired. Several important indicators of regional impairment are discussed. Several fish species that possess different core types and extents are discussed as potential indicators of regional impairment. Finally, I discuss the potential implications of regionally impaired watersheds as they pertain to biomonitoring, watershed restoration, and the management of fish species and communities. Although no single indicator of regional impairment exists, I propose that these complementary techniques provide a powerful framework to assess regional impairment.

Dedication

This document is a testament to love and support of my family. Without them, I would have never been able to undertake this incredible journey. My parents have done everything I could have ever asked. I will never be able to thank them enough for all of their love and support.

Acknowledgements

I wish to thank my committee members for their guidance and assistance through my graduate process. Each has provided valuable contributions to this document as well as my graduate education. I am indebted to Dr. J. Todd Petty, Dr. Kyle J. Hartman, Dr. Patricia M. Mazik, Dr. Michael P. Strager, and Dr. Stuart Welsh. I especially would like to thank Pat Mazik who agreed to fill Dr. Ron Fortney's position on my committee after he sustained injuries that forced his replacement. Ron was a valuable part of my graduate committee and I hope for his continued recovery. Additionally, Kyle Hartman deserves special recognition for his assistance and guidance with my Masters' research, which prepared me to undertake this project. Each person provided valuable contributions in their own way and all are greatly appreciated. These people have greatly improved the quality of my research, this document, and my time at West Virginia University.

The hands of numerous individuals are evident in this document as they have helped in some way along the journey. Instead of risking missing a name, I will thank all those that assisted with field work as a group. Many people helped collect data over three summers. They often endured grueling conditions and long hours. However, they all made the process of collecting data much more enjoyable. I also need to thank the many graduate students at West Virginia University that helped me in one way or another. Many were instrumental in data collection, developing ideas, and assisting with analysis or the writing process. In addition to their assistance with my education, many students past and present became good friends, offered support, and made my time at West Virginia University a truly enjoyable experience. I am also indebted to the many landowners in Preston and Tucker Counties that allowed me to access streams through their lands.

In particular, I would like to thank Roy Martin and Jennifer (Barker) Fulton. These two individuals were instrumental in much of the data collection. Roy led much of the fish and habitat data collection contained in this document. Without his assistance and leadership, I'd likely still be in the field collecting data. Jen collected all of the water quality data contained in this document. Additionally, she was instrumental in helping organize data, the lab's technicians, and graduate students. She was also often had a hand in maintaining my sanity.

Last but certainly not least, I owe many thanks to Todd Petty. He has truly been a wonderful mentor and I feel fortunate to have been able to learn from him and share ideas and aspirations with him. Aside from his assistance with my education and this research, he has been a valuable friend and mentor. I have benefited greatly from his assistance in many aspects of my life.

Local and Regional Impairment of Fish Assemblages in a Mined Appalachian Watershed

Jason G. Freund

Abstract	ii
Dedication	iii
Acknowledgements	iv
List of Tables	vii
List of Figures.....	ix
Executive Summary: Local and Regional Impairment of Fish Assemblages in a Mined Appalachian Watershed	1
Local and Regional Influences on Stream Fishes	1
Spatial Models and Stream Fishes	4
Cheat River as a Model System	6
Framework for Regional Effects Due to Local Impairment	7
Objectives and Summary of Results	8
Implications for Assessing Impairment	11
Biotic Response to AMD Treatment.....	12
Prioritizing AMD Restoration.....	13
Specific Restoration Targets	14
References	17
Chapter 1: Response of Fish and Macroinvertebrate Bioassessment Indices to Specific Stressor Levels in a Mined Appalachian Watershed	23
Abstract	23
Methods.....	27
Study Site	27
Water and Habitat Quality	28
Statistical Analysis.....	30
Results.....	33
Discussion	36
References.....	45
Figure Legend	55
Chapter 2: Effects of Local Impairment at a Watershed Scale: Examining the Effects of Acid Mine Drainage in the Cheat River Watershed.....	59
Abstract	59
Introduction.....	61
Methods.....	70
Study Area	70
Sampling Design.....	71
Physical Habitat Data.....	72
Water Chemistry Sampling.....	73
Stream Size and Spatial Position	74
Fish Data Collection	74
Statistical Analyses	77
Results.....	82
Utility of PCA in Partitioning Covariation among Variables.....	82

Assessing AMD-Impairment Categories	83
General Trends in Fish Assemblages.....	84
Species Richness Models.....	85
Species Presence/Absence Models	86
Species Core/Periphery Models.....	89
Discussion.....	91
Upper Cheat River Basin as a Reference Watershed.....	92
Local and Regional Impairment in the Lower Cheat River Basin.....	97
Mechanisms of Regional Impairment.....	100
Assessing Regional Impairment	102
Fish Species as Indicators of Regional Impairment.....	106
Implications of Regional Impairment	110
Implications for Watershed Restoration	112
Conclusions.....	114
References.....	115
Figure Legend	145
CURRICULUM VITAE.....	157

List of Tables

Chapter 1

Table 1. Means, coefficient of variation (%), and minimum and maximum values for habitat, water chemistry, WV SCI and MAHA-IBI data for the Cheat River watershed.49

Table 2. Results from principal component analysis on water chemistry data for the Cheat River basin. Factor loadings $> |0.4|$ are presented.....50

Table 3. Summary of regression analyses among habitat and water chemistry vs. macroinvertebrate (SCI), fish (IBI), and the combined SCI+IBI multimetric indices. Regression coefficients (R^2) are presented. Bold values signify significant tests at the $\alpha = 0.05$ (adjusted $\alpha = 0.0035$). Differences among regression coefficients are presented to indicate the direction and strength of differences among multimetric indices and abiotic factors.....51

Table 4. Discriminant function analysis (DFA) classification/misclassification analysis on ecological condition categories for WV SCI and MAHA-IBI data. Discriminant function analysis placed sites into categories according to their water chemistry. Each cell represents the number and percent classified into each combination.....52

Table 5. Linear regression coefficients for SCI vs. transformed abiotic factors. Impairment level is the level at which the independent variable correlates with an impaired SCI score (< 60.6). Impairment levels were not calculated for non-significant relationships.....53

Chapter 2

Table 1. Principal component analysis of Cheat River watershed water chemistry data.....127

Table 2. Occurrence of the most common species in the upper and lower basins of the Cheat River. For each the upper, lower, and entire Cheat River basin, the number of sampled stream reaches the species occurred in, the percentage of reaches it occurred in, and the relative rank for the number of sites the species occurred within is given. There were 27 upper basin sites, 37 lower basin sites, and 27 lower basin site with fishes present.....128

Table 3. Mean site proportion of tolerant and intolerant species (McCormick et al. 2001) for each site in the upper and lower basins as well as lower basin sites that contained fish.....129

Table 4. Discriminant function analysis classification/misclassification analysis using model built based on upper basin presence/absence data applied to lower basin to predict species occurrence. Percent values represent percentage of each present or absent species that were correctly or incorrectly classified within the lower basin.....	130
Table 5. Variables included in stepwise discriminant models developed from upper basin site data.....	132
Table 6. Number of correctly predicted presences, predicted presences, and percent of correct predictions from upper basin devived presence/absence models for all lower basin sites, unimpaired and moderately impaired lower basin sites, and upper basin sites.....	133
Table 7. Variables included in stepwise discriminant models developed from lower basin site data.....	134
Table 8. Discriminant function analysis classification/missclassification results for lower basin sites using models developed from the lower basin.....	135
Table 9. Variables included in stepwise discriminant models developed from unimpaired lower basin sites.....	137
Table 10. Variables included in stepwise discriminant models developed from moderately impaired lower basin sites.....	138
Table 11: Discriminant function analysis classification/missclassification results for lower basin models developed for unimpaired and moderately impaired sites within the lower basin.....	139
Table 12. Basin area (km ²) at the 25 th , 50 th , and 75 th percentile of relative cummulative proporiton of individuals for commonly occuring fish species in the upper Cheat River basin, their core extent of their core range (difference between 25 th and 75 th percentiles).....	141
Table 13. Percentage of sites in the lower and upper basins within the core and periphery area ranges where each commonly occuring species was present. Upper – lower basin represents the difference in the percentage of core and periphery sites that were occupied by each species.....	142
Table 14. Number and percent of core and periphery sites occupied for commonly occuring species within non-AMD impacted sites in the lower Cheat River basin compared to upper basin core and periphery occupancy. Upper basin core and periphery occupancy rates are provided for comparison.....	143
Table 15. Summary of core types, ecological attributes and representative species in the Cheat River watershed.....	144

List of Figures

Chapter 1

Figure 1. Adjusted WV SCI (A.) and adjusted MAHA-IBI (B.) scores against the AMD gradient (PC1).....	56
Figure 2. Plot of adjusted WV SCI, adjusted MAHA-IBI, and adjusted combined SCI + IBI against sulfate.....	57
Figure 3. Adjusted MAHA-IBI plotted against adjusted WV SCI score. Perfect agreement would result in a 1:1 ratio.....	58

Chapter 2

Figure 1. Fish sample sites in the upper and lower basin of the Cheat River, West Virginia.....	145
Figure 2. Water chemistry PCA factor scores for the Cheat River Basin.....	146
Figure 3. Stream size (log basin area) and water chemistry PC1 for upper and lower basin sites.....	147
Figure 4. Species richness model developed in the upper basin applied to upper basin data.....	148
Figure 5. Application of the upper basin species richness model in the lower basin. Lower basin sites are coded to their level of AMD-impairment and distance from a mainstem river (i.e., Cheat River or Big Sandy Creek).....	150
Figure 6. Ratio of observed and expected species richness from the upper basin species richness model as a function of first water chemistry PCA factor.....	151
Figure 7. Observed commonly occurring species richness as a function of expected common species richness determined from upper basin discriminant function presence/absence models.....	152
Figure 8. Relative cumulative proportion (Y1) and density (Y2) of individuals of commonly occurring species as a function of basin area. The relative cumulative proportion for each species (observed) is compared to the theoretical expected distribution under the null hypothesis that individuals accumulate in proportion to the amount of stream area (m ²) sampled.....	153

Executive Summary: Local and Regional Impairment of Fish Assemblages in a Mined Appalachian Watershed

State and federal agencies are charged with monitoring, maintaining, and restoring aquatic resources. These programs focus upon local impairments, largely ignoring how local impairments may have indirect, regional impacts. Currently, a framework to assess regional impairment does not exist within state and federal Clean Water Act mandated programs. Although the fish based Index of Biotic Integrity (IBI) may be responsive to regional conditions (Karr 1981, 1991), it relies inherently upon comparing local conditions with localized reference conditions limiting its responsiveness to regional impairment. Fishes provide an excellent long-term indicator of regional conditions since members of the taxon possess a wide range of trophic levels, longevity, and mobility. Thus, fish assemblages may be responsive to disturbances within the watershed (Karr 1981, 1991). A better understanding of how regional impairment and spatial position impact fish assemblages and multimetric indices of biotic integrity are needed to more accurately assess biotic conditions. Additionally, restoration activities that take into account regional factors such as position within the stream network and potential sources of colonizing individuals are more likely to have biologically significant results.

Local and Regional Influences on Stream Fishes

Local environmental variables are often strongly correlated with local species richness and the presence or absence of individual species (e.g. Sheldon 1968, Lotrich 1973, Schlosser 1982, Peterson and Rabeni 2001). However, much of the noise in these relationships may be explained by large-scale phenomenon. For the purposes of this

study, local refers to variables that are measured at the sample location such as habitat complexity, riparian condition, alkalinity, and drainage area. Regional influences are those processes occurring outside of the sample site that may be responsible for controlling local conditions such as barriers to dispersal (Winston et al. 1991, Ensign et al. 1997, Warren and Pardew 1998, Porto et al. 1999), distance of sample site from a source of colonizing individuals (Gorman 1986, Osborne and Wiley 1992, Schlosser 1995, Taylor 1997, Snodgrass and Meffe 1999, 2000), and upstream processes (Vannote et al. 1980, Naiman et al. 1987, Munn and Meyer 1990) that may influence local conditions.

Local and regional factors interact to structure fish assemblages in lotic ecosystems (Angermeier and Winston 1998). Species richness is often linked to basin area (Sheldon 1968, Angermeier and Schlosser 1989), habitat quality and complexity (Gorman and Karr 1978, Gorman 1988, Pinder and Morgan 1995), hydrologic variability (Kinsolving and Bain 1993, Poff and Allen 1995), water chemistry (Welsh and Perry 1997, Baldigo and Lawrence 2000), and spatial position relative to a species pool (Gorman 1986, Osborne and Wiley 1992). Additionally, biotic factors such as intraspecific and interspecific competition (Mendelson 1975, Gatz 1979, Grossman et al. 1998), predation (Schlosser 1987, Fraser et al. 1995), and prey abundance (Schlosser and Ebel 1989, Donalson and Nisbet 1999) also interact with abiotic factors in determining species richness (Meffe 1984, Guvrevitch et al. 2000, Taniguchi and Nakano 2000).

Regional factors that influence fish assemblages and species distributions are often neglected, but represent a growing research topic. In a series of influential papers, Osborne and Wiley (1992) detected greater species richness and higher index of biotic

integrity scores (Osborne et al. 1992) in streams located near a large regional species pool than were found in headwater tributaries. Spatial arrangement of habitat patches, occurring at many scales, can greatly affect the persistence of species and the overall community (Holyoak 2000).

Our understanding of the importance of connectivity and dispersal ability of fishes in the structuring of fish assemblages has greatly improved recently. Isolation of fishes above a barrier may lead to local extirpation (Winston et al. 1991, Fausch and Young 1995, Thurow et al. 1997, Warren and Pardew 1998) or reduce or eliminate potential complementary or supplementary habitats such as spawning grounds (Thurow et al. 1997, Beechie and Bolton 1999). Isolation that creates small habitat patches that may make populations more susceptible to reduction in genetic variability and stochastic events (Fausch and Young 1995, Donaldson and Nisbet 1999, Hilderbrand and Kershner 2000). Until recently, resident stream fishes were viewed as relatively immobile (Gerking 1953), however recent examinations have discovered that a segment of most populations are mobile and dispersal plays a critical role in the maintenance of stream fish populations (Gowan et al. 1994, Smithson and Johnston 1999). Barriers to dispersal and isolation interact to produce impacts on stream fish populations and assemblages (e.g. Winston et al. 1991, Fausch and Young 1995).

Two complementary areas of lotic research, beaver-influenced landscapes and isolated stream pools, support the importance of spatial position and dispersal in structuring fish assemblages. Beaver ponds, acting as a flow refugia, generally supply individuals to nearby lotic stream reaches (Schlosser and Ebel 1989, Schlosser 1995, Snodgrass and Meffe 1999, Schlosser and Kallenmeyer 2000) in a similar manner to how

large rivers supply tributaries with species (Gorman 1986, Osborne and Wiley 1992). Similar ecological processes are evident in temporal stream pools (Taylor 1997, Lonzarich et al. 1998, Matthews and Robison 1998). Common to these research avenues is the importance of the interaction among connectivity, dispersal, and spatial scale in structuring stream fish assemblages, species presence or absence, and relative abundance of individual species (Labbe and Fausch 2000, Fausch et al. 2002, Scheurer et al. 2003).

Spatial Models and Stream Fishes

The MacArthur and Wilson (1967) model, one of the most important and pervasive concepts in ecology, captures the relative roles of isolation and dispersal on maintenance of biotic communities. The model was the first to link island (patch) size and distance from a source of colonizing individuals to patterns of species richness. The basic form of the model proposes that larger islands and islands closer to the mainland have more species and that variability is greatest in small islands that are far from the mainland. This model is easily and widely applied to non-island systems. In terrestrial systems, the basic tenant is that patch size and the arrangement of patches on the landscape are important in structuring biotic assemblages. Underlying this basic model is the importance of dispersal in the maintenance of assemblages (Dunning et al. 1992, Stacey and Taper 1992).

Most spatial models are predicated on MacArthur and Wilson's (1967) Theory of Island Biogeography. Metapopulation models (Dunning et al. 1992, Hanski 1994) are useful in understanding patch occupancy and genetic consequences of dispersal. Source-sink dynamics (Pulliam 1988) account for differences in patch suitability. Highly suitable patches are likely to produce more individuals than the patch can maintain ($\lambda >$

1) and act as a source of individuals for sink habitats where the intrinsic rate of population growth is less than one. Habitat complementation and supplementation models (Dunning et al. 1992, Schlosser and Angermeier 1995) are robust and largely conceptual, but see Pope et al. (2000) for a quantitative application. A complementary habitat is a patch within the home range of an individual that provides a better reproductive, foraging, or refuge habitat than other suitable patches. A supplementary habitat is similar to a complementary habitat except that it is outside of the normal range that an individual sustains itself and the supplementary habitat is not sufficient for the individual to complete its life cycle. For example, brook trout in small tributaries may use a larger stream as a foraging supplement, but are unable to sustain a population in the larger stream as spawning areas are limited in larger streams.

Fish assemblages in lotic ecosystems are difficult to fit to existing spatial and patch models due to the linear nature of streams. However, these models can be important in characterizing patterns in species richness, abundance, and presence for stream fishes. The species-area relationship (Angermeier and Schlosser 1989) or the similar species-volume relationship (Taylor 1997) generally provides a robust predictor of species richness. Proximity to a source of colonizing individuals may increase species richness (Sheldon 1968, Osborne and Wiley 1992) and inclusion of a spatial term in species-area models may improve model fit. Our understanding of the importance of spatial position on stream fish assemblages continues to be developed. However, the influence of spatial position is often highly variable and may be condition specific (Grenouillet et al. 2004). Additionally, barriers in more typical patch models increase travel costs (Bernstein et al. 1991), but in linear stream networks barriers may eliminate

dispersal (Ensign et al. 1997, Porto et al. 1999) and increase the potential of population isolation (Winston et al. 1991, Warren and Pardew 1998).

Cheat River as a Model System

The Cheat River watershed provides a unique opportunity to test fish community structure; however the ecological processes that produce regional impairment in the Cheat River watershed are not unique. Local impairments that affect the core area of distribution and demographic processes for individual fish species or reduce connectivity may lead to regional impairment. Many anthropogenic sources of impairment such as channelization, dams, urbanization, chemical and wastewater inputs, and acid deposition may cause regional impairment.

The Cheat River is part of the upper Ohio River basin and is formed by the confluence of Shavers Fork and Black Fork Rivers in Parsons, WV. From the confluence of the Cheat River, it flows approximately 135 km before entering the Monongahela River near Point Marion, PA. The Cheat River drains an area of approximately 3,700 km², almost entirely within West Virginia. Coal bearing geology underlies much of the northern portion of the basin. Coal mining was historically important and remains a vital part of the local economy today. Acid mine drainage (AMD) commonly occurs where high sulfur coal was removed and water is able to interact with bacteria, oxygen, and pyrite to create a solution low in pH and high in sulfates and dissolved metals (Herlihy et al. 1990).

For the purposes of this research, the Cheat River basin was divided into upper and lower (upstream and downstream, respectively) basins. The lower basin, the northern most portion of the watershed, includes the area downstream of Pringle Run, the

first major, untreated acid mine impacted tributary. Many streams within this area are directly impacted by AMD and those not directly impacted may display altered fish assemblages due to the degradation of the lower watershed. Streams within the upper basin are unimpacted by AMD but experience the same anthropogenic disturbances as those in the lower basin (acid deposition, development, agriculture, and silviculture). A portion of the upper basin is within the Monongahela National Forest, providing the best opportunity for reference streams within the Cheat River watershed.

Framework for Regional Effects Due to Local Impairment

A framework to assess regional impairment can be developed by incorporating our knowledge of the relative roles of local and regional factors and how life history characteristics of individual stream fishes are expected to interact with these factors. Acid mine drainage produces two important conditions that have the potential to create regional impairment of stream fishes. First, local species richness and fish populations are generally reduced downstream of AMD inputs creating a reduction in the source of potential colonizing species and individuals. Secondly, AMD that isolates and reduces the effective patch size of lotic habitats have the potential to produce local extirpation of fish species. Together, interactions between these two factors are expected to increase the effects that either individual mechanism may have individually. Regional effects of AMD should be evident through reduction of biological integrity of the region, reduction of species richness, and predictable missing species compared with non-AMD impacted stream reaches in the lower basin.

Acid mine drainage in the Cheat River provides a uniquely severe source of impairment but mining drainage is pervasive in the Appalachian Mountains region and is

locally important throughout the world. Approximately 10% of stream reaches within the Northern Appalachian subregion of the National Stream Survey (NSS) are acidic due to AMD (Herlihy et al. 1990) and an unknown number of stream reaches may be indirectly impacted by AMD through regional consequences of AMD. Other common aquatic stressors such as water withdrawal, channelization, urbanization, and impoundment are likely to have similar potential regional impacts.

Objectives and Summary of Results

- 1. Quantify the range of variability of water chemistry, physical habitat, and stream size and spatial position within the Cheat River watershed.*

Large differences among water chemistry parameters were evident in the upper and lower basins. Water chemistry in the upper basin was generally in good or excellent condition and rarely exceeded water quality standards though some acid precipitation and development related effects were evident. One stream, Clay Lick Run, is among those on the WVDEP's 303d list for water quality impairment, likely due to sedimentation (WVDEP 1996). In contrast to the upper basin, many streams in the lower basin are moderately or severely impacted by acid mine drainage and other anthropogenic stressors. Principal component analysis provided evidence that AMD provided the dominant source of impairment. Water chemistry was much more variable in the lower basin with several sites showing no differences in comparison to upper basin sites. No identifiable patterns in habitat quality were evident except for the gradient in habitat quality. Geology within the watershed is variable (Cardwell et al. 1968). The upper basin is largely shale with limited amounts of limestone and sandstone whereas the lower basin is dominated by sandstone with limited amounts of

shale and limestone (WVDEP 1996). Sites were selected to best represent the range of stream sizes and proximity to a species pool. In the upper basin, the Cheat River was the only large river (5th order Strahler or larger). In the lower basin, both the Cheat River and Big Sandy Creek were considered to be mainstem rivers (Osborne and Wiley 1992). Additionally, Cheat Lake in the lower basin provided a source of unique lentic fishes that are absent or uncommon in the Cheat River watershed.

2. *Assess biotic response to the range of variability of abiotic factors.*

Multimetric indices effectively captured biotic response to physical, chemical, and spatial factors present in the Cheat River Watershed. Macroinvertebrate Stream Condition Index (SCI) was more responsive to most individual water chemistry parameters, particularly those indicative of AMD. Fish-based Index of Biotic Integrity (IBI) also effectively captured local water chemistry conditions but regional influences in IBI score were also evident. Macroinvertebrates appear to be better indicators of local conditions whereas fish assemblages may provide more information about regional condition, isolation, and spatial and temporal features.

Significant numbers of fish species were absent from sites in the lower Cheat River basin where they were expected to occur based on discriminant function models and the core/periphery view developed from upper basin sites. Local water quality degradation and degradation to core range explained many species' absences in lower basin streams. Non-tolerant species were commonly absent in moderately impaired streams and fishes were absent from all severely impacted streams. In the lower basin many large and moderate core species were absent from non-impaired stream reaches. Northern hogsucker (*Hypentelium nigricans*), central stoneroller

(*Campostoma anomalum*), smallmouth bass (*Micropterus dolomieu*), and river chub (*Nocomis micropogon*) were notably absent from many sites where they were predicted to be present. These species were most commonly present in streams near the Cheat River or Big Sandy Creek and largely absent from other subwatersheds.

Deviation in the species-area relationship is evident in the lower basin largely due to AMD impairment. However, sites that were not directly impaired by AMD also deviated from patterns observed in the upper basin. In the upper basin, stream size explained most of the pattern in species richness. Spatial position (link order difference) and water chemistry added some predictive power to the species richness model. For models developed in the lower basin, water chemistry explained much more of the variability in species richness than did water chemistry in the upper basin.

3. *Evidence of Regional Impairment*

A single measure of regional impairment is not conclusive. However a body of evidence is provided from inference from the bioassessment scores, the core-periphery model, models for individual species presence/absence, and the species richness model. When this information is coupled with life history characteristics of fish species, a view of regional impairment becomes evident. Other patterns add to the body of evidence which suggests a regional influence of AMD. Much greater than expected species richness in two lower basin sites that provide clean water refuge habitat for fishes in the moderately impaired and thermally-impacted Cheat River. In this case greater than expected species richness may be tied to regional impairment. This counterintuitive increase in richness, I hypothesize is due to an increased utilization of complementary and supplementary habitats in response to

highly variable water chemistry suitability. Probably the most telling symptom of regional impairment is the loss of particular species from sites within the basin area range they are expected to occupy. Isolated (e.g. Muddy Creek) and recovering watersheds (e.g. Little Sandy Creek) are devoid of several species that are expected to occur. However, barriers, either chemical or physical, and degraded regional conditions may prevent or slow the response of these watersheds to restoration activities.

Implications for Assessing Impairment

Fish assemblages provided information about regional scale phenomenon that could not be ascertained using water chemistry or macroinvertebrate assemblages. Water chemistry, benthic macroinvertebrates, and fishes each offer unique advantages and disadvantages in assessing biotic conditions. Water chemistry sampling, particularly when sites are sampled repeatedly, provided a valuable measure of the variability in water quality (Petty and Barker 2004). Water chemistry sampling is relatively inexpensive and time efficient, particularly over limited representative sample periods (Barbour et al. 1999). Water chemistry sampling is mandated by the Clean Water Act in assessment procedures and regulations are typically set based on concentrations of water quality parameters (WVDEP 1996). However, water samples can be temporally variable (e.g. Petty and Barker 2004) and single samples may be ineffective in capturing the range of variability in water chemistry parameters. Additionally, the response of biota is not implicit from water chemistry data.

Macroinvertebrates relate local conditions, water chemistry in particular, with biotic response. Macroinvertebrate sampling is inexpensive, efficient, and requires little

time and labor commitment in the field. Macroinvertebrates respond to short term habitat and water chemistry impairment and their high diversity correlates with an ability to indicate water chemistry over a range of sources of impairment (Barbour et al. 1999). Macroinvertebrate sampling requires extensive lab work and provides a local measure of water chemistry and habitat, but fails to address regional conditions.

Fishes add information about regional and long-term conditions that water chemistry and macroinvertebrates can not provide. Additionally, fish are highly visible and economically important and many surveys can be conducted in conjunction with state and federal sport fishing surveys. However, fish sampling is labor intensive and requires relatively expensive equipment. Sampling season is limited and samples at different seasons may not be comparable (Barbour et al. 1999). If local conditions are of primary interest, other indicators may be more effective.

Few Clean Water Act mandated bioassessment programs include fish sampling, but adding a limited fish component may provide information that is missing from many bioassessment programs. I suggest that fish sampling be prioritized to include sites that may display evidence of regional impairment and that sampling is coordinated with state and federal agencies when possible. Additionally, fish may be more effective than water chemistry or macroinvertebrates in assessing meaningful response to restoration activities. However, it should be noted that fish response to restoration may require five years before populations respond (Hunt 1976).

Biotic Response to AMD Treatment

Biotic response to improved local conditions is dependent upon regional species pool, connectivity and dispersal ability (Fisher 1990, Ensign et al. 1997, Taylor 1997,

Lonzarich et al. 1998). Fishes should be used as the biological target of restoration activities as they are economically important and represent the top predators in most aquatic systems (Barbour et al. 1999). Additionally, fishes will provide a measure of long-term sustainability that may not be evident through water chemistry or macroinvertebrate sampling.

Prioritizing AMD Restoration

To be successful, restoration needs to incorporate regional processes in addition to improving local conditions. Understanding the role of connectivity and dispersal of stream fishes and incorporating this knowledge in the prioritization of restoration is expected to lead to regional as well as local restoration of fish communities. Restoration of isolated stream reaches that does not improve connectivity may fail to improve fish species richness but populations of individual species may respond to improved water quality conditions. However, restoration activities that restore connectivity and improve conditions in moderate to large streams are likely to have the greatest regional benefit. Barriers, either physical or chemical, that prevent or reduces dispersal may make the reintroduction of fishes (Hilderbrand 2002, Schmetterling 2003). Reintroduction may be especially important for species that do not generally disperse widely such as sculpin (Petty and Grossman 2004) or when species are extirpated from the near-local species pool.

The core-periphery view described in detail in Chapter 2 provides a useful framework for assessing the range of stream basin areas that individual fish species commonly occur within. Restoration of individual fish populations may require that a significant proportion of the core range is intact for the species to persist. Additionally,

species are more likely to be found in peripheral areas when the core is intact.

Restoration should concentrate on reconnecting isolated stream reaches and restoring core habitat for moderate and large core species which make up the majority of species in the Cheat River watershed (Freund Chapter 2).

Specific Restoration Targets

Roaring Creek enters the Cheat River near Albright, WV, and has limited water quality degradation from Lick Run and other unknown sources of AMD. Water chemistry in Lick Run and Roaring Creek downstream of Lick Run is temporally variable and may be episodically toxic to stream fishes. Fish assemblages in lower Roaring Creek are relatively diverse and greatly exceeded model expectations generated from the non-AMD impacted upper basin. However, just downstream of Lick Run the fish assemblage is dominated by tolerant species and brook trout density is greatly reduced when compared to the upstream site on Roaring Creek. Additionally, this source of AMD may potentially limit species richness upstream by acting as a semi-permeable barrier to dispersal (Schlosser 1995). Treatment of Lick Run has the potential to restore native brook trout and other non-tolerant species in Roaring Creek.

The Little Sandy Creek watershed is moderately impacted by AMD, but is a recovering watershed (WVDEP 1996). Several species of fish were absent from fish collections, but were expected to be present. A physical barrier near the mouth of Little Sandy Creek may limit or eliminate dispersal of species from Big Sandy Creek.

Transplanting species from Big Sandy Creek or other nearby rivers may speed the recovery of Little Sandy Creek. Additionally, addressing a few small sources of AMD would limit potentially toxic events. In upper Beaver Creek, a small headwater tributary

of Little Sandy Creek, the West Virginia Division of Natural Resources (WVDNR) and a local Trout Unlimited (TU) chapter transplanted brook trout from a nearby stream.

Sampling after the first year indicated a range of age classes including numerous age-0 brook trout. Without this transplant, brook trout were unlikely to recolonize upper Beaver Creek as conditions in lower Beaver Creek coupled with a lack of nearby brook trout populations made recovery unlikely without assistance from the WVDNR and TU.

Muddy Creek is the single largest source of acidity and toxic metals to the Cheat River (WVDEP 1996, Williams et al. 1999). Martin Creek and the abandoned T&T mine are responsible for much of the water quality degradation. Treatment of small sources of AMD existing upstream of Martin Creek could improve ecological function to approximately 10 km of stream. Restoration of fish species in Muddy Creek would require fishes to be transplanted from nearby streams. Of particular note, northern hogsucker, central stoneroller, and river chub are absent from the Muddy Creek watershed. Additionally, Jump Rock Run is an acid precipitation impacted stream (WVDEP 1996) in the upper Muddy Creek watershed. Limestone treatment (Clayton et al. 1998) of Jump Rock Run would likely improve both Jump Rock Run and upper Muddy Creek. Recovery of fishes downstream of Martin Creek may be a long-term goal but currently heavy metal flocculants armor the streambed and likely have long-term consequences (Herlihy et al. 1990).

Restoration of the Cheat River is the ultimate goal of most involved in treating AMD in the Cheat River watershed. The single largest source of AMD is Muddy Creek which enters near the Canyon section, a popular whitewater rafting destination. Upstream, the first major sources of untreated AMD are Pringle, Lick, Morgan, and

Heather Runs. They enter the relatively unimpacted Cheat River from the west and create a localized reduction in species richness and biotic condition. Two other tributaries, Greens Run and Bull Creek enter the Cheat River upstream and downstream of the Canyon, respectively. Some of these tributaries may not be biologically recoverable, but reduction of acidity and heavy metals entering the Cheat River would improve biotic conditions. Additionally, targeting specific smaller watersheds as discussed above would slowly act to improve water chemistry in the Cheat River.

Acid mine drainage is difficult and expensive to treat (WVDEP 1996), but recovery of fisheries and renewed interest in the whitewater and tourism industry in the Cheat River watershed would bring additional money into the local economy. Because of the economic and social value of fishes, restoration activities should ultimately focus on the restoration of the Cheat River fish community. Biological endpoints such as fish index of Biotic Integrity (IBI, Karr 1981) and Stream Condition Index (SCI, WVDEP 1996) are necessary to evaluate the response of important biota to restoration and in setting goals for restoration activities. Restoration strategies that incorporate the effects of local conditions on regional quality are most likely to be effective in restoring fish assemblages at the watershed scale.

References

- Angermeier, P. L. and I. J. Schlosser. 1989. Species-area relationship for stream fishes. *Ecology* 70:1450-1462.
- Angermeier, P. L. and M. R. Winston. 1998. Local vs. regional influences on local diversity in stream fish communities of Virginia. *Ecology* 79:911-927.
- Baldigo, B. P. and G. B. Lawrence. 2000. Composition of fish communities in relation to stream acidification and habitat in the Neversink River, New York. *Transactions of the American Fisheries Society* 129:60-76.
- Barbour, M. T., J. Gerritsen, B.D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.
- Beechie, T. and S. Bolton. 1999. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds. *Fisheries* 24:6-15.
- Bernstein, C., A. Kacelnik, and J. R. Krebs. 1991. Individual decisions and the distribution of predators in a patchy environment. II. The influence of travel costs and structure of the environment. *Journal of Animal Ecology* 60:205-225.
- Cardwell, D. H., R. B. Erwin, H. P. Woodward, and C. W. Lotz. 1968. Geologic map of West Virginia. West Virginia Geologic and Economic Survey.
- Clayton, J. L., E. S. Dannaway, R. Menendez, H. W. Rauch, J. J. H. Renton, S. M. Sherlock, and P. E. Zurbuch. 1998. Application of limestone to restore fish communities in acidified streams. *North American Journal of Fisheries Management* 18:347-360.
- Donalson, D. D. and R. M. Nisbet. 1999. Population dynamics and spatial scale: effects of system size on population persistence. *Ecology* 80:2492-2507.
- Dunning, J. B., B. J. Danielson, and H. R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. *Oikos* 65:169-175.
- Ensign, W. E., K. N. Leftwich, P. L. Angermeier, and C. A. Dolloff. 1997. Factors influencing stream fish recovery following a large-scale disturbance. *Transactions of the American Fisheries Society* 126:895-907.
- Fausch, K. D. and M. K. Young. 1995. Evolutionarily significant units and movement of resident stream fishes: a cautionary tale. *American Fisheries Society Symposium* 17:360-370.

- Fausch, K. D., C. E. Torgersen, C. V. Baxter, and H. W. Li. 2002. Landscapes to riverscapes: Bridging the gap between research and conservation of stream fishes. *BioScience* 52:483-498.
- Fisher, S. G. 1990. Recovery processes in lotic ecosystems: limits of successional theory. *Environmental Management* 14:725-736.
- Fraser, D. F., J. F. Gilliam, and T. Yip-Hoi. 1995. Predation as an agent of population fragmentation in a tropical watershed. *Ecology* 76:1461-1472.
- Gatz Jr., A. J. 1979. Community organization in fishes as indicated by morphological features. *Ecology* 60:711-718.
- Gerking, S. D. 1953. Evidence for the concepts of home range and territory in stream fishes. *Ecology* 34:347-365.
- Gorman, O. T. and J. R. Karr. 1978. Habitat structure and stream fish communities. *Ecology* 59:507-515.
- Gorman, O. T. 1986. Assemblage organization of stream fishes: the effect of rivers on adventitious streams. *The American Naturalist* 128:611-616.
- Gorman, O. T. 1988. An experimental study of habitat use in an assemblage of Ozark minnows. *Ecology* 69:1239-1250.
- Gowan, C., M. K. Young, K. D. Fausch, and S. C. Riley. 1994. Restricted movement in resident steam salmonids: a paradigm lost. *Canadian Journal of Fisheries and Aquatic Sciences* 51:2626-2637.
- Grossman, G. D., R. E. Ratajczak, Jr., M. Crawford, and M. C. Freeman. 1998. Assemblage organization in stream fishes: effects of environmental variation and interspecific interactions. *Ecological Monographs* 68:395-420.
- Grenouillet, G., D. Pont, and C. Hérissé. 2004. Within-basin fish assemblage structure: the relative influence of habitat versus stream spatial position on local species richness. *Canadian Journal of Fisheries and Aquatic Sciences* 61:93-102.
- Guvrevitch, J., J. A. Morrison, and L. V. Hedges. 2000. The interaction between competition and predation: a meta-analysis of field experiments. *American Naturalist* 155:435-453.
- Hanski, I. 1994. A practical model of metapopulation dynamics. *Journal of Animal Ecology* 63:151-162.
- Helfman, G. S., B. B. Collette, and D. E. Facey. 1999. *The diversity of fishes*. Blackwell Science, Ltd. Malden, Massachusetts. 528 pages.

- Herlihy, A. T., P. R. Kaufmann, M. E. Mitch, and D. D. Brown. 1990. Regional estimates of acid mine drainage impact on streams in the Mid-Atlantic and Southeastern United States. *Water, Air, and Soil Pollution* 50:91-107.
- Hilderbrand, R. H. and J. L. Kershner. 2000. Conserving inland cutthroat trout in small streams: how much stream is enough? *North American Journal of Fisheries Management* 20:13-520.
- Hilderbrand, R. H. 2002. Simulation supplementation strategies for restoring and maintaining stream resident cutthroat trout populations. *North American Journal of Fisheries Management* 22(3):879-887.
- Holyoak, M. 2000. Habitat patch arrangement and metapopulation persistence of predators and prey. *The American Naturalist* 156: 378-389.
- Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6:91-107.
- Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management. *Ecological Applications* 1:66-84.
- Kinsolving, A. D. and M. B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. *Ecological Applications* 3: 531-544.
- Labbe, T. R. and K. D. Fausch. 2000. Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. *Ecological Applications* 10: 1774-1791.
- Lonzarich, D. G., M. L. Warren, Jr., and M. R. E. Lonzarich. 1998. Effects of habitat isolation on the recovery of fish assemblages in experimentally defaunated stream pools in Arkansas. *Canadian Journal of Fisheries and Aquatic Sciences* 55:2141-2149.
- Lotrich, V. A. 1973. Growth, production, and community composition of fishes inhabiting a first-, second-, and third-order stream of Eastern Kentucky. *Ecological Monographs* 43:377-397.
- MacArthur, R. and E. O. Wilson. 1967. *The theory of island biogeography*. Princeton University Press, Princeton, NJ.
- Matthews, W. J. and H. W. Robison. 1998. Influence of drainage connectivity, drainage area, and regional species richness on fishes of the interior highlands in Arkansas. *The American Midland Naturalist* 139: 1-19.
- Meffe, G. K. 1984. Effects of abiotic disturbance on coexistence of predator-prey fish species. *Ecology* 65:1525-1534.

- Mendelson, J. 1975. Feeding relationships among species of *Notropis* (Pisces: Cyprinidae) in a Wisconsin stream. *Ecological Monographs* 45:199-232.
- Munn, N. L. and J. L. Meyer. 1990. Habitat-specific solute retention in two small streams: an intersite comparison. *Ecology* 71:2069-2082.
- Naiman, R. J., J. M. Melillo, M. A. Lock, T. E. Ford, and S. R. Reice. 1987. Longitudinal patterns of ecosystem processes and community structure in a subarctic river continuum. *Ecology* 68:1139-1156.
- Osborne, L. L., S. L. Kohler, P. B. Bayley, D. M. Day, W. A. Bertrand, M. J. Wiley, and R. Sauer. 1992. Influence of stream location in a drainage network on the Index of Biotic Integrity. *Transactions of the American Fisheries Society* 121:635-643.
- Osborne, L. L. and M. J. Wiley. 1992. Influence of tributary spatial position on the structure of warmwater fish communities. *Canadian Journal of Fisheries and Aquatics Sciences* 49:671-681.
- Peterson, J. T., and C. F. Rabeni. 2001. The relation of fish assemblages to channel units in an Ozark stream. *Transactions of the American Fisheries Society* 130:911-926.
- Petty, J. T. and J. Barker. 2004. Water quality variability, trace metals and implications for restoring a mined Appalachian watershed. 2004 National Meeting of the American Society of Mining and Reclamation and the 25th West Virginia Surface Mining Drainage Task Force.
- Petty, J. T. and G. D. Grossman. 2004. Restricted movement by mottled sculpin (pisces: cottidae) in a southern Appalachian stream. *Freshwater Biology* 49:631-645.
- Pinder, M. J. and I. R. P. Morgan. 1995. Interactions of pH and habitat on Cyprinid distributions in Appalachian streams of Maryland. *Transactions of the American Fisheries Society* 124: 94-102.
- Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegard, B. D. Richter, R. E. Sparks, and J. C. Stomberg. 1997. The natural flow regime. *BioScience* 47:769-784.
- Pope, S. E., L. Fahrig, and H. G. Merriam. 2000. Landscape complementation and metapopulation effects on leopard frog populations. *Ecology* 81:2498-2508.
- Porto, L. M., R. L. McLaughlin, and D. L. G. Noakes. 1999. Low-head barrier dams restrict the movements of fishes in two Lake Ontario streams. *North American Journal of Fisheries Management* 19:1028-1036.
- Pulliam, H. R. 1988. Sources, sinks, and population regulation. *American Naturalist* 132:652-661.

- Scheurer, J. A., K. D. Fausch, and K. R. Bestgen. 2003. Multiscale processes regulate brassy minnow persistence in a Great Plains river. *Transactions of the American Fisheries Society* 132:840-855.
- Schlosser, I. J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs* 52:395-414.
- Schlosser, I. J. 1987. The role of predation in age- and size-related habitat use by stream fishes. *Ecology* 68:651-659.
- Schlosser, I. J. and K. K. Ebel. 1989. Effects of flow regime and cyprinid predation on a headwater stream. *Ecological Monographs* 59:41-57.
- Schlosser, I. J. 1995. Dispersal, boundary processes, and trophic-level interactions in streams adjacent to beaver ponds. *Ecology* 76:908-925.
- Schlosser, I. J. and P. L. Angermeier. 1995. Spatial variation in demographic processes of lotic fishes: conceptual models, empirical evidence, and implications for research. *American Fisheries Society Symposium* 17:392-401.
- Schlosser, I. J. and L. W. Kallenmeyn. 2000. Spatial variation in fish assemblages across a beaver-influenced successional landscape. *Ecology* 81:1371-1382.
- Schmetterling, D. A. 2003. Reconnecting a fragmented river: Movements of Westslope cutthroat trout and bull trout after transport upstream of Milltown Dam, Montana. *North American Journal of Fisheries Management* 23:721-731.
- Sheldon, A. L. 1968. Species diversity and longitudinal succession in stream fishes. *Ecology* 49:193-198.
- Smithson, E. B. and C. E. Johnston. 1999. Movement patterns of stream fishes in a Ouachita Highlands stream: an examination of the restricted movement paradigm. *Transactions of the American Fisheries Society* 128:847-853.
- Snodgrass, J. W. and G. K. Meffe. 1999. Habitat use and temporal dynamics of blackwater stream fishes in and adjacent to beaver ponds. *Copeia* 1999:628-639.
- Snodgrass, J. W. and G. K. Meffe. 2000. Influence of beavers on stream fish assemblages: effects of pond age and watershed position. *Ecology* 79:928-942.
- Stacey, P. B. and M. Taper. 1992. Environmental variation and the persistence of small populations. *Ecological Applications* 2:18-29.
- Taniguchi, Y. and S. Nakano. 2000. Condition-specific competition: implications for the altitudinal distribution of stream fishes. *Ecology* 81:2027-2039.

- Taylor, C. M. 1997. Fish species richness and incidence patterns in isolated and connected stream pools: effects of pool volume and spatial position. *Oecologia* 110:560-566.
- Thurrow, R. F., D. C. Lee, and B. E. Rieman. 1997. Distribution and status of seven native salmonids in the interior Columbia River basin and portions of the Klamath River and Great Basins. *North American Journal of Fisheries Management* 17: 1094-1110.
- Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37:130-137.
- Warren, M. L. Jr., and M. G. Pardew. 1998. Road crossings as barriers to small-stream fish movement. *Transactions of the American Fisheries Society* 127:637-644.
- Welsh, S. A. and S. A. Perry. 1997. Acidification and fish occurrence in the upper Cheat River Drainage, West Virginia. *Journal of the American Water Resources Association* 33:423-429.
- Williams, D. R., M. E. Clark, and J. B. Brown. 1999. Stream water quality in coal mined areas of the lower Cheat River basin, West Virginia and Pennsylvania, during low-flow conditions, July 1997. *Water Resource Investigations Report* 98-4258 8.
- Winston, M. R., C. M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. *Transactions of the American Fisheries Society* 120:98-105
- WVDEP. 1996. An ecological assessment of the Cheat River watershed. West Virginia Division of Environmental Protection. Report 0502004 – 1996.

Chapter 1: Response of Fish and Macroinvertebrate Bioassessment Indices to Specific Stressor Levels in a Mined Appalachian Watershed

Abstract

Multimetric indices based on fish and benthic macroinvertebrate assemblages are commonly used to assess the biological health of aquatic ecosystems. However, the response of biological indices to specific stressor levels is rarely known. I quantified the relative response of a fish-based index of biotic integrity (IBI) and a benthic macroinvertebrate-based index to acid mine drainage related stressors in the Cheat River watershed, West Virginia. Fish and benthic macroinvertebrate assemblage data (% taxon composition) were used to calculate Mid-Atlantic Highland Assessment Index of Biotic Integrity (MAHA-IBI) and West Virginia Stream Condition Index (WV SCI) scores for each site. Both MAHA-IBI and WV SCI scores were scaled by dividing all scores by the highest value observed in the watershed. Water chemistry was dominated by constituents indicative of AMD. Both fish and macroinvertebrate assemblages responded to water chemistry parameters and principal component analysis (PCA) factor scores that summarized covariance among water chemistry parameters. Macroinvertebrate assemblages were more responsive to chemical stressors from AMD than were fish assemblages. However, in many cases information from macroinvertebrate and fish multimetric indices provided complementary information, presumably because fish assemblages are responsive to regional conditions, and invertebrates are more responsive to local conditions. Results suggest that monitoring programs and prioritization of mine discharge treatment need to consider biological response to mining-related water chemistry impairment.

Keywords: Bioassessment; Cheat River, West Virginia; Index of Biotic Integrity;
Macroinvertebrates; Stream Condition Index; Trace Metals;

Introduction

Multimetric bioassessment indices are predicated on the principle that biotic communities are predictable, stable, and indicative of historic and current stressors (Karr 1981, Matthews 1986, McCormick et al. 2001). Both benthic macroinvertebrate and fish assemblages are commonly used as bioindicators in state and federal aquatic monitoring programs (USEPA 2002). These indices have several important values. First, they can be refined to meet local conditions and stressors (Shields et al. 1995, Lyons et al. 1996, McCormick et al. 2001, Snyder et al. 2003, Weigel 2003). Second, they provide standardized data collection and analysis procedures (Barbour et al. 1999, Yoder and Smith 1999). Third, they are easily adapted to meet regulatory and monitoring program needs (Yoder and Smith 1999). Multimetric indices are widely used by regulatory agencies to estimate current biotic conditions, determine biotic community response to environmental stressors, to assist in regulatory decision making, and for the long-term monitoring of aquatic ecosystems.

Both macroinvertebrate and fish assemblages have been shown to be responsive to variability in physical habitat (Hawkins et al. 1982, Shields et al. 1995, Snyder et al. 2003, Weigel 2003, Pirhalla 2004), warm- and cold-water streams (Shields et al. 1995, Lyons et al. 1996), and water chemistry (Wallace et al. 1996, Snyder et al. 2003, Weigel 2003). The ability of benthic macroinvertebrate and fish multimetric indices to respond to physical and chemical stressors in the aquatic environment is paramount to their competence as diagnostic and regulatory tools (Karr 1981). It is often argued that macroinvertebrates are good indicators of local conditions because they are diverse and display a wide range of tolerance to physical and chemical stressors (Hilsenhoff 1982,

Barbour et al. 1999, Sloane and Norris 2003, Weigel 2003). Also, benthic macroinvertebrates are represented by many different trophic levels (Barbour et al. 1999). Because fishes are highly mobile and tend to be longer lived, they may be better indicators of historic and chronic stressors, the effects of isolation, and stressors which have regional impacts in addition to local impairment (Karr 1981, Barbour et al. 1999).

The West Virginia Stream Condition Index (WV SCI) and the Mid-Atlantic Highland Index of Biotic Integrity (MAHA-IBI) were developed to assess biotic conditions in West Virginia watersheds. The West Virginia Division of Environmental Protection (WVDEP) uses the West Virginia Stream Condition Index (Tetra Tech 2000) based on macroinvertebrate assemblages in the agency's Clean Water Act mandated watershed assessments (e. g. WVDEP 1996). The United States Environmental Protection Agency (US EPA) developed a fish-based index of biotic integrity as part of the Mid-Atlantic Highlands Assessment that included the entire state of West Virginia (McCormick et al. 2001).

Although multiple anthropogenic stressors impact most watersheds (Karr 1981, Barbour et al. 1996), mining-related discharge can overwhelm other impacts (WVDEP 1996, Woodward et al. 1997, Sloane and Norris 2003, Petty and Barker 2004). Mine-related discharge is high in trace metals (Woodward et al. 1997, Williams et al. 1999, Petty and Barker 2004) and can greatly affect macroinvertebrate (WVDEP 1996, Sloane and Norris 2003) and fish (Woodward et al. 1997) assemblages. Pyrite-rich coal fields in the Appalachians produce acid mine drainage (AMD), which in addition to elevated heavy metal concentrations, is high in sulfate and low in pH, alkalinity, and hardness (WVDEP 1996, Williams et al. 1999, Petty and Barker 2004). Streams that are

moderately impacted by AMD have water chemistry that is temporally variable and receive pulses of trace metals (Petty and Barker 2004). Acid mine drainage also reduces stream acid neutralizing capacity thereby increasing the stream's susceptibility to acid deposition (Pinder and Morgan 1995, Welsh and Perry 1997).

Despite a comprehensive understanding of the chemical characteristics of AMD impacted streams, the precise nature of fish and invertebrate response to AMD remains uncertain. To assess the biological impact of AMD in the Cheat River basin, my objectives were to: 1) quantify stream-to-stream variation in water chemistry within the Cheat River basin, particularly as they relate to mining activity; 2) assess the relative ability of fish and benthic macroinvertebrate based multimetric indices to respond to gradients in water chemistry; 3) determine the response of multimetric indices to specific chemical constituents; and 4) determine the extent to which combined information from fish and macroinvertebrates communities can be used to guide management decisions in AMD watersheds.

Methods

Study Site

The Cheat River watershed is a major tributary to the Monongahela River, and drains an area of 3,678 km² within West Virginia (WVDEP 1996). The Cheat River is formed by the confluence of the Black and Shavers Forks and flows approximately 100 km until the river is impounded by the Lake Lynn dam near the West Virginia-Pennsylvania border. Tributaries to the Cheat River tend to be dendritic, but vary greatly in water and habitat quality. Approximately 43 km downstream of the river's origin, the Cheat River receives several important sources of acid mine drainage. This point, in

conjunction with a shift in bedrock geology, effectively separates the upper and lower Cheat River basins. Both upper and lower basins were dominated by sandstone and shale geology with limited outcrops of Greenbrier limestone. The lower basin, however, possesses extensive coal deposits. Principal mined seams include the Upper and Lower Freeport and Bakerstown, both of which have high pyrite coal. In contrast, the upper basin possess few coal deposits and not of them have been mined historically (Cardwell et al. 1968).

Water and Habitat Quality

Forty-three sites within the Cheat River watershed were selected across a range of stream size, geology, position within a drainage network, and mining intensity. Water quality impairment in the Cheat River basin is dominated by AMD, but potentially includes other anthropogenic stressors such as acid precipitation and development.

Water quality data were collected in April 2003 to mirror the manner in which the WVDEP and MAHA-IBI data were collected. Additionally, early spring coincided with the poorest water quality in moderately impaired sites (Petty and Barker 2004).

Temperature (C), pH, specific conductance ($\mu\text{S}/\text{cm}^3$), dissolved oxygen (mg/l) and total dissolved solids (mg/l) were measured using a YSI 650 with a 600 XL sonde (Yellow Springs, Ohio). A 500 ml sample was filtered using a Nalgene polysulfone filter holder and receiver using mixed cellulose ester membrane 0.45 μm pore size disk. Filtered samples were treated with 5 ml 1:1 nitric acid to bring the pH below 2 to prevent dissolved metals from falling from solution. Filtered samples were used for analysis of aluminum, iron, manganese, nickel, cadmium, chromium, and calcium and total hardness (mg/l). A 1 liter grab sample was collected for analysis of alkalinity, acidity, and sulfate.

Unfiltered samples were kept on ice after collection and stored in the laboratory at 4°C until analysis could be completed. All laboratory analysis were conducted at Black Rock Test Lab in Morgantown, WV.

Rapid visual habitat assessment (RVHA) data were collected for each site following USEPA protocols (Barbour et al. 1999). Two consistent observers were used to estimate these parameters to minimize ambiguity in the data (Roper and Scharnecchia 1995). Rapid visual habitat assessment (RVHA) percent score was used for statistical analyses.

Macroinvertebrate WV SCI scores were obtained from the West Virginia Division of Environmental Protection (WVDEP). Macroinvertebrate sites were sampled as part of WV DEP watershed assessment data collection during 1996, 1999, 2001, and 2002. The most recently sampled site was selected and all benthic macroinvertebrate sites corresponded with sites sampled for physical habitat, water chemistry, and fish assemblages. For the purposes of this study, stream reach was defined as a section of stream bounded by tributaries of first order or larger from 1:24:000 scale West Virginia GIS coverage. West Virginia Stream Condition Index scores were calculated by the WVDEP following protocols set forth by (Tetra Tech 2000). The WV SCI is a composite of six individual metrics which each use different aspects of macroinvertebrate assemblage structure. Imbedded in the final WV SCI score is taxon richness; Ephemeroptera (E), Plecoptera (P), and Trichoptera (T) richness; percent EPT; percent tolerant; percent dominance; and a modified Hilsenhoff biotic index (HBI, Hilsenhoff 1988). The final score was scaled to vary from 0 to 100.

Fish sampling protocols were modified slightly from EPA-EMAP protocols outlined by McCormick et al. (2001). Stream reaches were 40 times the mean stream width (MSW) or a minimum of 150 m and a maximum of 300 m in length (Lyons 1992, Yoder and Smith 1999). Sampling greater distance would not yield any significant improvements in IBI score or species richness (Ohio EPA 1989; Freund, unpublished data). Protocols specified that shocking effort was not to exceed 1800 seconds; measured as the amount of time in seconds that electrical current is applied to the stream (Ohio EPA 1989). Upstream and downstream boundaries were blocked with a large net if a natural barrier did not exist. One to three backpack electrofishing units were employed with a portable seine net which captured stunned fishes not collected by electrofishing operators or assisting net handlers. Fishes were processed and recorded separately for each 50 m section. Fish were identified to species and returned to the stream. Voucher specimens were collected and preserved in a 10% buffered formalin solution in 2002 or 95% ethyl alcohol in 2003. Fish IBI scores were calculated according to McCormick et al. (2001) with metric scoring equations provided by the United States Environmental Protection Agency (USEPA) based on reference sites for the MAHA-IBI (McCormick et al. 2001).

Statistical Analysis

Macroinvertebrate WV SCI scores and fish IBI scores were scaled relative to the highest scores received for the watershed and is referred to henceforth as adjusted WV SCI and adjusted MAHA-IBI, respectively. Scaling resulted in a proportion, which could be used in statistical comparisons and allowed for unbiased comparisons between WV SCI and MAHA-IBI. Percent and proportion data were arc sine square root transformed

to approximate normality before inclusion in statistical analyses and water chemistry values other than pH were natural log +1 transformed to improve normality (Zar 1999). Both unadjusted WV SCI and MAHA-IBI scores were classified into condition categories (Excellent, Good, Fair, and Poor) according to the WVDEP (WVDEP 1996), and McCormick et al. (2001), respectively. Excellent and good were considered unimpaired whereas fair and poor were considered impaired (unadjusted score < 60.6; Tetra Tech 2000).

Principal component analysis (PCA) was used to organize water chemistry and habitat along gradients of covariation. I considered Eigenvalues greater than 1.0 and individual factor loadings greater than 0.4 to be significant (McGarigal et al. 2000). Discriminant function analysis (DFA) classification/misclassification was used to classify sites as upper or lower basin based on water chemistry and RVHA score.

Two statistical procedures were used to determine the response of multimetric indices to the variability in water chemistry and habitat variability. First, simple linear regression was used to test if the macroinvertebrate and fish multimetric indices and the combined macroinvertebrate and fish index were responsive to each of the significant PCA axes. Second, principal component gradients were entered with total RVHA score as independent variables in stepwise multiple linear regression models for WV SCI, MAHA-IBI, and the combined index. Principal components were determined for each basin individually and were used in stepwise multiple linear regression models for each basin separately.

I used simple linear regression to examine the response of a combined macroinvertebrate and fish index to water quality. The sequence of tests were identical to

those used on the WV SCI and MAHA-IBI data separately. A Kruskal-Wallis test on ranked correlation coefficients for WV SCI, MAHA-IBI, and the combined index was used to determine if the combined index was statistically different from the individual indices.

Relationships among fish and macroinvertebrate indices as well as the combined index and individual water quality parameters and RVHA score were tested for significance. Simple linear regression on transformed data was used to test for relationships among multimetric indices and habitat or individual water chemistry parameters. To control for multiple pair wise errors, the *a priori* alpha level (0.05) was divided by the overall number of regression tests on WV SCI and MAHA-IBI data (Zar 1999). The resultant critical p-value was 0.0035 ($=0.05/14$). To assess the adjusted IBI and SCI score correlation at each site, a Wilcoxon paired-sample test (Zar 1999) was used.

Finally, I assessed the level at which each independent variable from the above WV SCI regression equations produced an impairment. To assess at what level the independent variable resulted in impairment; I used the regression equations developed above to determine when the impairment level ($\text{WV SCI} < 60.6$) was reached. This was only done when RVHA score and individual water chemistry parameters that were significantly related to WV SCI score. The level at which macroinvertebrate assemblages were never impaired was determined as the highest or lowest value of each independent variable above or below which never resulted in an impaired SCI score. Conversely, the level of each stressor at which macroinvertebrate assemblages were

always impaired was the level of each independent variable above or below which all sites were impaired.

Results

The impacts of mining on water chemistry in the Cheat River watershed are pervasive (Table 1). Elevated total acidity, sulfate, specific conductance, total dissolved solids, and heavy metal concentrations along with decreased alkalinity and pH in the lower basin correspond with extensive sources of AMD. Water chemistry in the upper basin indicated a lack of coal mining-related impacts.

Principal component analysis of water chemistry data produced three significant axes (Table 2). The first principal component represented a gradient where large positive values were indicative of increasing AMD impairment, whereas negative factor scores characterized sites with increased pH, hardness, and alkalinity. The second principal component represented a hardness and alkalinity gradient: positive factor scores represented sites with relatively high pH, alkalinity, calcium and total hardness, whereas negative scores characterized area of high total acidity and total aluminum. The third axis was uninterpretable and not considered further. Sites within the upper basin exhibited low variability on PC1 but high variability on PC2. In the lower basin both PC1 and PC2 factor scores were highly variable. Non-mining impaired lower basin sites were generally similar to upper basin sites.

Principal components one and two along with RVHA scores were used as independent variables in stepwise multiple linear regression models to explain how these factors influenced WV SCI, MAHA-IBI, and a combined metric score. In the WV SCI model, only PC1, the AMD-related gradient, remained in the final model producing a

linear R^2 value of 0.83 (Figure 1A). The model for MAHA-IBI also retained only PC1 (partial $R^2 = 0.51$; Figure 1B). Using PC1 separated into upper and lower basins did not produce a significant model for the upper basin. Models for the lower basin were similar to the model for the entire Cheat River basin except that habitat explained about 4.7% of the variance in IBI score in the lower basin. The second principal component (PC2) showed weak correlation with either adjusted SCI or IBI.

Both macroinvertebrate WV SCI and fish MAHA-IBI indices were correlated with most water chemistry parameters (Table 3). However, adjusted WV SCI and MAHA-IBI were relatively weakly correlated (Figure 2; $R^2 = 0.53$, $p < 0.0001$). West Virginia stream condition index was more strongly correlated, based on R^2 values, with water chemistry parameters indicative of AMD (Fe, Al, Ni, sulfate) whereas MAHA-IBI scores were more strongly correlated with alkalinity (Table 3). Sulfate was most strongly correlated with both WV SCI (Figure 3) and MAHA IBI scores (Table 3). Aluminum, manganese, and nickel (Table 3) also showed strong ($R^2 > 0.6$) correlation with WV SCI score. The strongest relationships among water chemistry parameters and MAHA-IBI were sulfate, alkalinity, and manganese (Table 3). Alkalinity was the only water chemistry parameter that exhibited a stronger relationship with the fish index than the macroinvertebrate index (Table 3). The greatest differences in correlation coefficients of macroinvertebrates and fishes between SCI and IBI involved iron, nickel, and aluminum concentrations (Table 3). A Wilcoxon paired-sample test of R^2 values provided additional evidence that water chemistry parameters were more highly correlated with the macroinvertebrate index ($T^+ = 93$, $T^- = 21$, $df = 14$, $p = 0.005$).

Multivariate analysis supports the previous univariate results indicating that water chemistry is effective in structuring macroinvertebrate and fish assemblages and corresponding multimetric index values. Discriminant function analysis produced strong classification into WV SCI and MAHA-IBI categories based on water chemistry parameters (Table 4). Stepwise DFA indicated MAHA-IBI categories were categorized based on total hardness, alkalinity, and nickel. The WV SCI model based on DFA retained total dissolved solids, alkalinity, calcium hardness, and chromium.

Combining adjusted WV SCI and MAHA-IBI into a single average index rarely produced stronger correlations with water chemistry values than did either the WV SCI or the MAHA-IBI indices by themselves (Table 3). Over the host of multimetric index and water chemistry correlations, the combined index showed no improvement over either the WV SCI or MAHA-IBI indices (Kruskal-Wallis; $F_{2,39} = 2.04$, $p\text{-value} = 0.1439$). Stepwise multiple linear regression using PC1, PC2, PC3 and RVHA score as independent variables produced a final model explaining 76% of the variation in the combined index. Only PC1, the AMD-gradient, was retained in the final model. The final model compares favorably to the IBI only model R^2 of 0.51 but not the SCI model which had an R^2 of 0.83.

Specific water chemistry parameters were often linked to impairment of the macroinvertebrate community (Table 5). Sulfate was most strongly related to SCI score and impairment was detected at 54.4 mg/l. Impairment linked to nickel concentrations occurred at the lowest concentration. Most water chemistry constituents had discernable levels above or below which impairment either always or never occurred. However,

some constituents did not have discernable levels where impairment or lack of impairment was evident.

Discussion

Mining-related impacts in the lower Cheat River basin are pervasive. Multimetric indices using benthic macroinvertebrate or fish assemblages as indicators of physical and water chemistry aquatic stressors were responsive to stressors in the Cheat River watershed as has been shown in other watersheds (e.g. McCormick et al. 2001, USEPA 2002, Snyder et al. 2003, Weigel 2003). Indices using macroinvertebrate and fish assemblages, as well as the combined index, were all responsive to water chemistry. Over the range of variability in water chemistry in the Cheat River watershed, macroinvertebrates were more responsive to individual water chemistry parameters except for alkalinity, a measure of potential productivity (Kwak and Waters 1997). Macroinvertebrates, which displayed greater response to water chemistry, are commonly used in most bioassessment programs, whereas fishes are not commonly utilized in bioassessment programs (USEPA 2002).

State (WV SCI; Tetra Tech 2000) and regional (MAHA-IBI; McCormick et al. 2001) indices were used since more specific indices were not available. The MAHA-IBI was developed for portions of Pennsylvania, Maryland, Virginia, and the entire state of West Virginia and was developed to be responsive to a multitude of aquatic stressors (McCormick et al. 2001). However, mining-related impacts are among the most biologically devastating anthropogenic stressors (WVDEP 1996, Williams et al. 1999) and the MAHA-IBI responded well to mining impacts. Development of a Cheat River

watershed specific IBI was hindered by a lack of large, unimpacted reference sites, a problem common in the development of many IBIs (Simon 1999).

The WV SCI is utilized by all WVDEP watershed assessments and was developed to be responsive to stressors across West Virginia (WVDEP 1996; Tetra Tech 2000). The WVSCI is a general index, including metrics encompassed by bioassessment programs across the country (Barbour et al. 1999, Weigel 2003) indicating a responsiveness over a host of sources of environmental degradation. It is unlikely that a macroinvertebrate index developed specifically for the Cheat River watershed would vary substantially from the WVSCI model.

Since the effects of mining drainage on biotic communities are so pervasive, the development of bioassessment tools that are more specific to mining-related issues may not provide much additional information. Both the regional IBI and the state-wide SCI were responsive to mining-related impairment. Development of a Cheat River watershed specific index or reference conditions would likely produce negligible improvement and reduces the ability to compare the Cheat River to other watersheds across the state.

Surprisingly, I found only one peer reviewed article that directly compared macroinvertebrate and fish multimetric index response to identical conditions (Paller 2001) and few that incorporated both macroinvertebrates and fish into a single index (Kerans and Karr 1994, Whittier et al. 2002). In my study, the macroinvertebrate index showed greater correlation with individual water chemistry parameters and principal component scores indicating that WV SCI was more responsive to water quality impairment than was the fish IBI. Paller (2001) found little difference in the response of macroinvertebrate and fish indices; however differences in conclusions may be in part

due to benthic macroinvertebrate collection methods. WVDEP (1996) sampled with active gear (D-frame kick net) whereas Paller (2001) used passive gear (Hester-Dendy artificial substrates). Additionally, vast differences among high gradient, mining impacted streams and low gradient coastal floodplain streams confound comparisons between our studies.

Correlation between the adjusted macroinvertebrate and fish indices was significant, but was relatively weak for moderately impaired streams, similar to Paller (2001). Much of the deviation came from sites that lacked fishes, sites with less than 10 individual fishes were collected and an IBI score could not be calculated (McCormick et al. 2001), or had fish assemblages that were dominated by tolerant species that produced low IBI scores, but had relatively high corresponding adjusted macroinvertebrate SCI scores. Few if any fish species are able to tolerate pH and metal concentrations sampled in the severely impacted streams, macroinvertebrates are more numerically diverse and have more highly tolerant taxa (Barbour et al. 1999, Kilgour and Barton 1999).

There are several plausible explanations for disagreement between adjusted WV SCI and MAHA-IBI scores, most of which are site-specific. Most common were sites with low IBI and relatively higher SCI. These sites may be indicative of historic or regional impairment, because fish are longer lived (Karr 1981, 1991) and more susceptible to isolation effects (Lonzarich et al. 1998, Matthews and Robison 1998). Seldom did IBI score exceed SCI score and most sites that had higher IBI scores were located in smaller streams near a large stream or river that may have served as a source of colonizing individuals, resulting in elevated IBI scores (Osborne et al. 1992). This level of

disagreement provides evidence that using a single taxonomic group may not provide the depth of information needed by state and federal biomonitoring programs (Paller 2001).

The combined metric produced marginal improvement over the individual macroinvertebrate or fish indices. However, development of an index that incorporated both (e.g. Kerans and Karr 1994) may provide information lacking in the individual indices. Development of a more specific combined index which combines the advantages of local information which macroinvertebrates provide (Hilsenhoff 1982, Sloane and Norris 2003, Weigel 2003) and regional information which fishes assemblages can provide (Karr 1981, Osborne et al. 1992) should improve the response of a multimetric index to local and regional impacts of anthropogenic stressors. Keran and Karr (1994) incorporated characteristics of macroinvertebrate and fish assemblages into a single index within the Tennessee River valley.

A second, more commonly used approach is to develop separate indices for different taxonomic groups which produce complementary sources of information regarding biotic condition. This is the approach inherent to the US EPA's Environmental Monitoring and Assessment Program (EMAP; Barbour et al. 1999; Whittier et al. 2002).

Macroinvertebrates and fish along with periphyton each provide information at different spatial and temporal scales and are differentially responsive to stressors (Barbour et al. 1999). Incorporating knowledge of the mechanisms for both agreement and disagreement in indices can relay important information not provided by a single index.

Using PCA to incorporate the covariance in the water quality data set improved the correlation between water chemistry and bioassessment indices. The second PCA axis, which described a gradient of acid neutralizing components, provided information

that may augment the first axis, but by itself provides little predictive power. This is not surprising given the relatively small range and low alkalinity and hardness concentrations within the Cheat River watershed may not strongly influence macroinvertebrate or fish assemblages or production (Kwak and Waters 1997).

Water quality within the Cheat River watershed can effectively be captured by three categories; unimpaired, moderately impaired, and severely impaired (Petty and Barker 2004). Severely degraded sites were limited to the lower basin and are characterized by large inputs of AMD, greatly reduced macroinvertebrate assemblages, and a lack of fish. Moderately impacted sites showed the greatest range of variability in both biotic communities as evidenced by multimetric index scores and water chemistry parameters (Petty and Barker 2004). All of the upper basin sites were characterized by water chemistry lacking an AMD signature. However, in the lower basin relatively few streams did not display an AMD signature. Our results agree with others that have examined the responses of biotic communities to AMD in the Cheat River basin (WVDEP 1996) as well as other mined watersheds (Herlihy et al. 1990). Water chemistry overwhelmed any effects that physical habitat might have on biotic communities in the Cheat River, at least as measured through RVHA protocols. Mining discharge can impact habitat through metal deposition which increases embeddedness (Williams et al. 1999). Despite this potential impact, RVHA scores were slightly higher in the lower basin which is likely related to geologic differences that are responsible for habitat formation (Panfil and Jacobson 2001).

Levels of individual chemical constituents were often linked with macroinvertebrate impairment. However, correlations should not be viewed as causation (Clements 2004).

Most metals had discernable concentrations below which macroinvertebrate SCI scores were never below impairment levels. These values can be considered to be the maximum level of water chemistry impairment that is unlikely to result in a quantifiable response in macroinvertebrate communities. Across all streams, the level at which streams were never impaired was generally well below state and federal water quality criteria.

However, it is unlikely that a single individual chemical constituent is responsible for degraded macroinvertebrate communities. In field experiments conducted in mining drainage from Colorado that was heavy in zinc, copper, and cadmium, Clements (2004) found the greatest response of macroinvertebrates when all three constituents were present. Mining discharges are generally highly variable in the concentrations of toxic metals (Petty and Barker 2004). Although a particular constituent may not be present at a level that is likely to be toxic to macroinvertebrates, other constituents may be elevated. Consequently, a single chemical constituent is unlikely to be responsible for impairment through the Cheat River basin. Rather, elevated concentrations of several chemical constituents present in AMD are likely to be responsible for community-level impairment. However, state and federal regulations are set to monitor individual water chemistry parameters and may not be responsive to the interactive effects of chemical constituents that may all be below water quality standards.

The level at which impairment was set was a very conservative estimate (60.6) and is well below the level at which streams are classified into the “fair” range ($68 > \text{SCI} > 45$; Tetra Tech 2000). Many of the sites in the Cheat Basin fell within this “gray area” between 68 and 60.6 that are not considered to be impaired. Laboratory experiments to

detect levels of chemical constituents and combinations of water chemistry parameters that are likely to result in impairment are needed to support field-based observations.

Watershed assessment programs in mined watersheds need to consider assimilating fish into standard monitoring programs that have generally included only macroinvertebrates and water chemistry (WVDEP 1996, Williams et al. 1999). Water chemistry response to treatment may be highly variable (Petty and Barker 2004) and infrequent sampling may miss episodic events that often are responsible for structuring biotic communities (Baker et al. 1996). Both macroinvertebrate and fish assemblages provide spatial and temporal information which can not be gleaned simply from water chemistry data. However, fishes may provide evidence of regional conditions that macroinvertebrate are unable to provide. Disruption of connectivity may slow the recovery of fish assemblages (Lonzarich et al. 1998, Matthews and Robison 1998) whereas macroinvertebrate assemblages and water chemistry may not detect this form of impairment. Additionally, fish being a high visible member of aquatic communities and economically important (Karr 1981, Barbour et al. 1999), they represent a unique and important endpoint in assessing the effectiveness of restoration activities.

Given that water chemistry is the overriding factor limiting both macroinvertebrate and fish assemblages in the Cheat River watershed and that most of the degradation is due to mining-related activities, reducing acidity and toxic metals from mining discharges is an obvious first step in the recovery of the lower basin. While there has been much research and experimentation concerning AMD treatment within the Cheat River basin (e.g. Skousen et al. 1998), little work has addressed biotic response to mine discharge treatment. While this research avenue provides valuable information

concerning the assessment of biotic communities and their response to water chemistry degradation, dose-response curves for individual species and chemical constituents are needed to set goals for treatment. In addition, using multivariate methods are necessary to capture the covariation among water chemistry measures in mining discharge (Petty and Barker 2004). Much as relying upon a single biotic taxon, measuring the success of AMD treatment based only upon individual chemical constituents, particularly pH, acidity, and alkalinity or hardness, may fail to capture toxic levels of metals that increase their solubility at higher pH levels (e.g. Poleo et al. 1994).

Prioritization of mining discharge treatment requires a broad knowledge-base incorporating people from diverse disciplines. Planners who prioritize mining-discharge treatment need to consider spatial and temporal biotic response in addition to water quality improvement. Restoration of connectivity is vital in restoration of fish assemblages (Detenbeck et al. 1992, Lonzarich et al. 1998) and may provide watershed-scale indirect impacts to fish assemblages by providing supplementary habitat to fishes (Schlosser 1987, Schlosser and Angermeier 1995). Unless treatment of mining discharge incorporates biological endpoints, improvement in biological conditions may not be realized.

Bioassessment tools such as the WV SCI and MAHA-IBI are useful tools in determining important biological responses to mining-related water chemistry degradation. Simply relying upon any single measure, whether it be water chemistry or a biological index, is unlikely to provide information needed across spatial and temporal scales (Barbour et al. 1999, Paller 2001). Macroinvertebrate and fish assemblages

provide complimentary information that is vital in understanding the spatial and temporal response of biota to water quality and habitat degradation or remediation.

References

- Angermeier, P. L. 2000. The natural imperative for biological conservation. *Conservation Biology* 14:373-381.
- Baker, P. P. and 11 coauthors. 1996. Episodic acidification of small streams in the Northeastern United States: effects on fish populations. *Ecological Applications* 6:422-437.
- Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish*, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.
- Clements, W. H. 2004. Small-scale experiments support causal relationships between metal contamination and macroinvertebrate community response. *Ecological Applications* 14:954-967.
- Detenbeck, N. E., P. W. DeVore, G. J. Niemi, and A. Lima. 1992. Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. *Environmental Management* 16:33-53.
- Hawkins, C. P., M. L. Murphy, and N. H. Anderson. 1982. Effects of Canopy, Substrate Composition, and Gradient on the Structure of Macroinvertebrate Communities in Cascade Range Streams of Oregon. *Ecology* 63(6):1840-1856.
- Herlihy, A. T., P. R. Kaufmann, M. E. Mitch, and D. D. Brown. 1990. Regional estimates of acid mine drainage impact on streams in the Mid-Atlantic and Southeastern United States. *Water, Air, and Soil Pollution* 50:91-107.
- Hilsenhoff, W. L. 1982. Using a biotic index to evaluate water quality in streams. Wisconsin Department of Natural Resources Technical Bulletin 132:22pp.
- Hilsenhoff, W. L. 1988. Rapid field assessment of organic stream pollution. *Journal of North American Benthological Society* 7(1):65-68.
- Keran, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications* 4(4):768-785.
- Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6:91-107.
- Kilgour, B.W. and D. R. Barton. 1999. Associations between stream fish and benthos across environmental gradients in southern Ontario, Canada. *Freshwater Biology* 41:553-566.

- Kwak, T. J. and T. F. Waters. 1997. Trout production dynamics and water quality in Minnesota streams. *Transactions of the American Fisheries Society* 126:35-48.
- Lonzarich, Jr., D. G., M. L. Warren, and M. R. E. Lonzarich. 1998. Effects of habitat isolation on the recovery of fish assemblages in experimentally defaunated stream pools in Arkansas. *Canadian Journal of Fisheries and Aquatic Sciences* 55:2141-2149.
- Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. *North American Journal of Fisheries Management* 12:198-203.
- Lyons, J., L. Wang, and T. D. Simonson. 1996. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. *North American Journal of Fisheries Management* 16:241-256.
- McGarigal, K., S. Cushman, and S. Stafford. 2000. *Multivariate Statistics for Wildlife and Ecology Research*. Springer-Verlag, New York, 283p.
- Matthews, W. J. 1986. Fish faunal structure in an Ozark stream: stability, persistence, and a catastrophic flood. *Copeia* 1986:388-397.
- Mathews, W. J., and H. W. Robison. 1998. Influence of drainage connectivity, drainage area, and regional species richness on fishes of the interior highlands in Arkansas. *The American Midland Naturalist* 139:1-19.
- McCormick, F. H., R. M. Hughes, P. R. Kaufmann, D. V. Peck, J. L. Stoddard, and A. T. Herlihy. 2001. Development of an index of biotic integrity for the Mid-Atlantic Highland Region. *Transactions of the American Fisheries Society* 130:857-877.
- Ohio Environmental Protection Agency. 1989. *Biological criteria for the protection of aquatic life. Vol. III. Users manual for biological field assessment of Ohio surface waters*. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, OH.
- Osborne, L. L., S. L. Kohler, P. B. Bayley, D. M. Day, W. A. Bertrand, M. J. Wiley, and R. Sauer. 1992. Influence of stream location in a drainage network on the index of biotic integrity. *Transactions of the American Fisheries Society* 121:635-643.
- Paller, M. H. 2001. Comparison of fish and macroinvertebrate bioassessments from South Carolina coastal plain streams. *Aquatic Ecosystem Health and Management* 4(2):175-186
- Panfil, M. A. and R. B. Jacobson. 2001. Relations among geology, physiography, land use, and stream habitat conditions in the Buffalo and Current River systems, Missouri and Arkansas. U. S. Geological Survey Biological Science Report 2001-0005.

- Petty, J. T., and J. Barker. 2004. Water quality variability, trace metals and implications for restoring a mined Appalachian watershed. Proceedings of the National Meeting of the American Society of Mining and Reclamation. ASMR, Lexington, KY.
- Pinder, M. J. and I. R. P. Morgan. 1995. Interactions of pH and habitat on Cyprinid distributions in Appalachian streams of Maryland. Transactions of the American Fisheries Society 124:94-102.
- Pirhalla, D. E. 2004. Evaluating fish-habitat relationships for refining regional indexes of biotic integrity: development of a tolerance index of habitat degradation for Maryland stream fishes. Transactions of the American Fisheries Society 133:144-159.
- Poleo, A. B. S., E. Lydersen, B. O. Rosseland, F. Kroglund, B. Salbu, R. D. Vogt, and A. Kvellestad. 1994. Increased mortality of fish due to changing Al-chemistry of mixing zones between limed streams and acid tributaries. Water, Air, and Soil Pollution 75:339-351.
- Roper, B. B. and D. L. Scharnecchia. 1995. Observer variability in classifying habitat types in stream surveys. North American Journal of Fisheries Management 15: 49-53.
- Schlosser, I. J. and P. L. Angermeier. 1995. Spatial variation in demographic processes of lotic fishes: conceptual models, empirical evidence, and implications for research. American Fisheries Society Symposium 17:392-401.
- Schlosser, I. J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303:71-81.
- Shields, Jr., F. D., S. S. Knight, and C. M. Cooper. 1995. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. Hydrobiologia 312:191-208.
- Simon, T. P. (editor). 1999. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Florida.
- Skousen, J. G., A. Rose, G. Geidel, J. Foreman, R. Evans, and W. Hellier. 1998. Handbook of Technologies for Avoidance and Remediation of Acid Mine Drainage, ADTI of the USOSM, Published by the National Mine Land Reclamation Center, WVU, Morgantown, WV.
- Sloane, P. I. W., and R. H. Norris. 2003. Relationship of AUSRIVAS-based macroinvertebrate predictive model outputs to a metal pollution gradient. Journal of the North American Benthological Society 22:457-471.

- Snyder, C. D., J. A. Young, R. Villella, and D. P. Lemarié. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. *Landscape Ecology* 18:647-664.
- Tetra Tech, Inc. 2000. A Stream Condition Index for West Virginia Wadeable Streams. Owings Mills, MD.
- USEPA. 2002. Summary of Biological Assessment Programs and Biocriteria Development for States, Tribes, Territories, and Interstate Commissions: Streams and Wadeable Rivers. EPA-822-R-02-048. U.S. Environmental Protection Agency.
- Wallace, J. B., J. W. Grubaugh, and M. R. Whiles. 1996. Biotic indices and stream ecosystem processes: results from an experimental study. *Ecological Applications* 6:140-151.
- Whittier, T. R., S. G. Paulsen, D. P. Larsen, S. A. Peterson, A. T. Herlihy, and P. R. Kaufmann. 2002. Indicators of ecological stress and their extent in the populations of Northeastern lakes: a regional-scale assessment. *BioScience* 52:235-247.
- Weigel, B. 2003. Development of stream macroinvertebrate models that predict watershed and local stressors in Wisconsin. *Journal of the North American Benthological Society* 22:123-142.
- Welsh, S. A. and S. A. Perry. 1997. Acidification and fish occurrence in the upper Cheat River Drainage, West Virginia. *Journal of the American Water Resources Association* 33:423-429.
- Williams, D. R., M. E. Clark, and J. B. Brown. 1999. Stream water quality in coal mined areas of the lower Cheat River basin, West Virginia and Pennsylvania, during low-flow conditions, July 1997. Water Resource Investigations Report 98-4258, 8 pages.
- Woodward, D. F., J. N. Goldstein, A. M. Farag, and W. G. Brumgaugh. 1997. Cutthroat trout avoidance of metals and conditions characteristic of a mining waste site: Coeur d' Alene River, Idaho. *Transaction of the American Fisheries Society* 126: 699-706.
- WVDEP. 1996. An ecological assessment of the Cheat River watershed. West Virginia Division of Environmental Protection. Report 0502004 – 1996.
- Yoder, C. O. and M. A. Smith. 1999. Using fish assemblages in a state biological assessment and criteria program: essential concepts and considerations. Pages 17-56 in T. P. Smith, editor. *Assessing the sustainability and biological integrity of water resources using fish communities*. CRC Press, Boca Raton, Florida.

Zar, J. H. 1999. Biostatistical Analysis: Fourth Edition. Prentice Hall, Upper Saddle River, New Jersey.

Table 1. Means, coefficient of variation (%), and minimum and maximum values for habitat, water chemistry, WV SCI and MAHA-IBI data for the Cheat River watershed.

	Sample Size	Mean	CV	Min. - Max.
pH	46	6.4	18	2.7 - 8.4
Specific Conductance ($\mu\text{S}/\text{cm}^3$)	46	240.0	185	31.0 - 2390.0
Alkalinity (mg/l CaCO_3 eq.)	43	7.8	74	0 - 23.1
Total Acidity (mg/l CaCO_3 eq.)	46	27.0	229	0 - 348
Sulfate (mg/l)	46	59.6	212	5.9 - 670.0
Al (mg/l)	46	1.4502	366	0.010 - 25.900
Cd (mg/l)	46	0.0032	261	0 - 0.045
Cr (mg/l)	46	0.005	271	0 - 0.074
Fe (mg/l)	46	2.03	457	0.01 - 58.55
Mn (mg/l)	46	0.327	338	0 - 7.384
Ni (mg/l)	46	0.020	231	0.001 - 0.249
Calcium Hardness(mg/l)	46	15.8	92	3.3 - 55.3
Total Hardness (mg/l)	46	41.7	144	7.3 - 379
RVHA % Score	41	76	19	38 - 98
WV SCI	49	73	26	14 - 93
MAHA IBI	49	51	55	0 - 84

Table 2. Results from principal component analysis on water chemistry data for the Cheat River basin. Factor loadings $> |0.4|$ are presented.

	PC1	PC2
pH	-0.86	.
Specific Conductance	0.86	.
Total Dissolved Solids	0.96	.
Alkalinity	-0.61	0.63
Total Acidity	0.63	-0.47
Sulfate	0.91	.
Al	0.93	-0.45
Cd	.	.
Cr	.	.
Fe	0.90	.
Mn	0.92	.
Ni	0.95	.
Calcium Hardness	0.62	0.68
Total Hardness	0.78	0.49
 % Variance Explained	 61%	 13%
Eigenvalue	8.56	1.87

Table 3. Summary of regression analyses among habitat and water chemistry vs. macroinvertebrate (SCI), fish (IBI), and the combined SCI+IBI multimetric indices. Regression coefficients (R^2) are presented. Bold values signify significant tests at the $\alpha = 0.05$ (adjusted $\alpha = 0.0035$). Differences among regression coefficients are presented to indicate the direction and strength of differences among multimetric indices and abiotic factors.

Independent Variable	Dependent Variable			Regression Coefficient Differences		
	SCI	IBI	SCI+IBI	SCI- IBI	SCI - (SCI+IBI)	IBI - (SCI + IBI)
RVHA	0.024	0.006	0.000	0.018	0.024	0.006
Alkalinity (mg/l CaCO ₃ eq.)	0.420	0.443	0.502	-0.023	-0.083	-0.059
Sulfate (mg/l)	0.705	0.547	0.698	0.157	0.007	-0.151
Specific Conductance ($\mu\text{S}/\text{cm}^3$)	0.509	0.396	0.505	0.113	0.004	-0.109
Acidity (mg/l CaCO ₃ eq.)	0.211	0.102	0.161	0.109	0.050	-0.059
Al (mg/l)	0.654	0.331	0.509	0.323	0.145	-0.178
Cd (mg/l)	0.041	0.015	0.027	0.025	0.014	-0.012
Cr (mg/l)	0.033	0.033	0.038	0.000	-0.005	-0.005
Mn (mg/l)	0.618	0.376	0.532	0.242	0.086	-0.156
Fe (mg/l)	0.573	0.227	0.393	0.346	0.180	-0.166
Ni (mg/l)	0.649	0.312	0.492	0.338	0.158	-0.180
Calcium Hardness (mg/l CaCO ₃ eq.)	0.346	0.233	0.315	0.113	0.030	-0.083
Total Hardness (mg/l CaCO ₃ eq.)	0.524	0.391	0.507	0.133	0.017	-0.116
pH	0.430	0.280	0.385	0.150	0.045	-0.105

Table 4. Discriminant function analysis (DFA) classification/misclassification analysis on ecological condition categories for WV SCI and MAHA-IBI data. Discriminant function analysis placed sites into categories according to their water chemistry. Each cell represents the number and percent classified into each combination.

WV SCI						
Classified From:		Classified Into:				
		Excellent	Good	Fair	Poor	Total
	Excellent	7 (70%)	3 (30%)	--	--	10
	Good	5 (29%)	11 (65%)	1 (6%)	--	17
	Fair	1 (9%)	1 (9%)	9 (82%)	--	11
	Poor	--	--	--	5 (100%)	5
MAHA-IBI						
Classified From:		Classified Into:				
		Excellent	Good	Fair	Poor	Total
	Excellent	1 (50%)	1 (50%)	--	--	2
	Good	--	6 (60%)	4 (40%)	0	10
	Fair	0	2 (14%)	12 (86%)	0	14
	Poor	1 (6%)	--	3 (18%)	13 (77%)	17

Table 5. Linear regression coefficients for SCI vs. transformed abiotic factors. Impairment level is the level at which the independent variable correlates with an impaired SCI score (< 60.6). Impairment levels were not calculated for non-significant relationships. Levels above or below which impairment was never evident are labeled never impaired. Conversely, levels above or below which all sites were impaired are labeled as always impaired.

Independent Variable	Intercept	Slope	R²	Never Impaired	Impairment Level	Always Impaired
RVHA	1.16	-0.23	0.024	.	.	.
Alkalinity (mg/l CaCO ₃ eq.)	48.62	13.12	0.420	> 11.0	< 1.5	N/A
Sulfate (mg/l)	117.25	-14.11	0.705	< 47.6	> 54.4	> 164.0
Specific Conductance (µS/cm ³)	131.35	-12.60	0.509	< 124.0	> 274.0	N/A
Acidity (mg/l CaCO ₃ eq.)	84.02	-5.71	0.211	N/A	> 59.4	> 235.0
Al (mg/l)	78.88	-21.09	0.654	< 0.070	> 1.38	> 6.80
Cd (mg/l)	73.51	-466.98	0.041	.	.	.
Cr (mg/l)	73.33	-291.44	0.033	.	.	.
Mn (mg/l)	79.57	-42.53	0.618	< 0.05	> 0.56	> 1.06
Fe (mg/l)	77.32	-18.38	0.573	< 0.02	> 1.48	> 5.32
Ni (mg/l)	79.02	-364.81	0.649	< 0.007	> 0.005	> 0.032
Calcium Hardness (mg/l CaCO ₃ eq.)	110.18	-15.08	0.346	< 12.7	> 25.8	N/A
Total Hardness (mg/l CaCO ₃ eq.)	123.09	-15.57	0.524	< 36.9	> 54.3	> 125
pH	0.67	11.09	0.430	N/A	< 5.4	< 3.1

Figure Legend

Figure 1. Adjusted WV SCI (A.) and adjusted MAHA-IBI (B.) scores against the AMD gradient (PC1).

Figure 2. Adjusted MAHA-IBI plotted against adjusted WV SCI score. Perfect agreement would result in a 1:1 ratio.

Figure 3. Plot of WV SCI, MAHA-IBI, and the combined SCI + IBI indices against sulfate.

Figure 1

Figure 2

Figure 3

Chapter 2: Effects of Local Impairment at a Watershed Scale: Examining the Effects of Acid Mine Drainage in the Cheat River Watershed

Abstract

Currently, stream ecological health assessments do not consider the potential impacts of local degradation that may be manifested at a regional scale. Although a fish based index of biotic integrity may detect regional impairments through reduced IBI scores, mechanisms to explain and develop these indices are typically based on local conditions such as water chemistry and physical habitat. The lower Cheat River basin, West Virginia, is impacted by acid mine drainage (AMD) whereas the upper basin is unimpacted by AMD and provides a useful reference system for developing expectations that can be applied and tested in the lower basin.

To examine the potential impact that acid mine drainage (AMD) has on fish assemblages in the Cheat River Watershed, I employed three complementary methods of assessing regional impacts. First, a multiple regression model was developed to relate physical, chemical, and spatial variables to fish species richness. The model described 79% of the variability in species richness in the upper basin. The model included basin area (partial $R^2 = 0.75$) and link order difference (partial $R^2 = 0.04$). Species richness in the lower basin was generally reduced from predicted values and the upper basin model when applied to the lower basin performed poorly ($R^2 = 0.34$). The model performed similarly in non-impaired ($R^2 = 0.52$) and moderately AMD-impaired ($R^2 = 0.56$) stream reaches. A species richness model developed for the lower basin included yielded an R^2 of 0.54 and included Shreve order (0.31), water quality PC1 (0.10), and link order difference (0.13). Second, core habitat area, defined as the range of basin area between

the 25th and 75th percentiles of each species' cumulative distribution of individuals, served as a conservative measure of critical basin area for species conservation. Lastly, discriminant function models were developed for commonly occurring upper basin species and applied to the lower basin. The core/periphery and presence-absence models both showed higher occupancy rates in upper basin sites compared to lower basin sites. Between these final two methods, absences of species that were expected to be present in the lower basin were established. Disruption of demographic processes or reduction of connectivity to the core range is expected to reduce the probability of species persistence and may have regional impact in peripheral areas. Cumulatively, these three methods which examined assemblage and individual fish species response provided a framework to assess the regional affects of local impairment.

I propose that while AMD provided a unique model system, the findings are not unique and may be prevalent in watersheds impacted by urbanization, channelization, acid precipitation, and other sources of chemical or physical impairment. As we continue to develop better understanding of watershed connectivity and how disruptions may manifest themselves at larger spatial scales, potential regional impairments should be considered in bioassessment monitoring, watershed restoration, and risk assessment.

Key Words: acid mine drainage (AMD); Cheat River, West Virginia; core/periphery model; fish assemblages; life history; local and regional factors; species-area relationship; stream fragmentation; watershed impairment

Introduction

Our view of stream ecology has recently expanded from a local scale view to a broader regional view (Fausch et al. 2002). The emerging view that local species richness and presence/absence of individual fish species are tied to large-scale ecological processes is largely a result of incorporating landscape ecology into stream fish ecology (Schlosser 1991, 1995a, 1995b, Schlosser and Angermeier 1995, Fausch et al. 2002). This dynamic landscape model incorporates fish life history characteristics, particularly the role of dispersal, within a landscape view that incorporates heterogeneity at multiple scales. Fundamental to this theoretical model is that local processes and observations are in part a function of the regional riverscape in which they exist (Fausch et al. 2002).

Our view of ecological impairment has been narrowly shaped. We tend to view impairment as local phenomenon while regional processes are largely ignored (Angermeier and Winston 1998, Fausch et al. 2002). As a result, stream ecosystem impairment and the administration of the Clean Water Act (FWPCA 1972) have been focused at the local, stream reach scale. Additionally, multimetric indices of ecological condition have been developed based upon correlations between biological communities and local physio-chemical conditions (Karr 1981, 1991, McCormick et al. 2001). However, influences of regional processes on stream fish assemblage structure are becoming more apparent (Jackson and Harvey 1989, Angermeier and Winston 1998). Nevertheless, regional influences are not currently addressed in most ecological assessment or management programs (Fausch et al. 2002).

Effective management of aquatic resources will require that we understand how local communities are influenced by the interaction of local and regional processes. Most

patterns in species richness can be explained by large-scale, regional processes (Cornell and Lawton 1992), and deviations from expected patterns are often hypothesized to be due to local variation in physical and chemical conditions (Angermeier and Winston 1998). The regional species pool is set through historical speciation events (Strange and Burr 1997). The regional pool may gain species through non-native species introductions (Rahel 2000, Scott and Helfman 2001) and lose species through extirpation (Baltz and Moyle 1993, Oberdorff et al. 1997). Local species diversity can be viewed as the interaction of individual fish species' life history characteristics within the regional species pool being passed through a succession of physical, chemical, and biological filters (Tonn et al. 1990). These "filters" may be local or regional in nature but more likely, they interact among spatial scales (Fausch et al. 2002).

The effects of local conditions and impairment on fishes are fairly well understood. Within the Cheat River basin, I expected fish species to respond to hydrologic variability (Poff and Allan 1995), acid precipitation (Welsh and Perry 1997), acid mine drainage (Herlihy et al. 1990), stream gradient (Fausch et al. 2002), and predation (Schlosser 1987) among other physical, chemical, and biological filters. Additionally, these factors are likely to be interactive (Meffe 1994, Schlosser and Ebel 1989, Grossman et al. 1998, Taniguchi and Nakano 2000), forming complex but predictable fish assemblages.

Drainage area generally provides a robust predictor of fish species richness (Sheldon 1968, Angermeier and Schlosser 1989, Newall and Magnuson 1999). Drainage area is strongly correlated with physical, chemical, and biotic gradients that structure aquatic communities (Vannote et al. 1980). Following the river continuum concept

(Vannote et al. 1980), fish species richness is expected to follow a similar pattern of increasing downstream species richness observed in benthic macroinvertebrates (Lotrich 1973, Vannote et al. 1980, Matthews and Robison 1998, Parsons et al. 2003). The species-basin area relationship provides a framework to examine the relative roles of biotic and abiotic factors as well as their interaction in explaining the variability in fish species richness (Angermeier and Schlosser 1989, Matthews and Robison 1998).

Many of the patterns observed in fish species presence and density may be explained by water temperature (Taniguchi et al. 1998, Torgersen et al. 1999, Taniguchi and Nakano 2000, Wehrly et al. 2003). Deviations in the thermal profile such as coldwater inputs through spring flow, upwelling through alluvial deposits, and tributary confluences may supply warmwater streams with localized coldwater refugia (Bilby 1984, Torgersen et al. 1999) thereby increasing species diversity.

Fish species vary greatly in their susceptibility to water quality degradation, and resultant fish assemblages may be indicative of current and historic water chemistry (Karr 1991). Fish assemblages have been shown to respond to water chemistry degradation as a result of anthropogenic disturbances, such as acid precipitation (DeWalle et al. 1988, Welsh and Perry 1997, Baldigo and Lawrence 2000), urban and residential runoff (Wear et al. 1998, Wang et al. 2000), agricultural practices (Karr 1981, Osborne and Wiley 1988), and mine drainage (Woodward et al. 1997, Freund Chapter 1). Natural water chemistry variation is tied to watershed climate, lithology, and geochemistry (Minshall et al. 1983). Nutrient retention and uptake are spatially variable (Munn and Meyer 1990) and have been linked to fish species presence (Guegan et al. 1998), most notably several species of omnivorous cyprinids (Felly and Hill 1983) and catostomids (Curry and Spacie

1984). Species richness (Lotrich 1973) and production (Waters et al. 1993, Kwak and Waters 1997) has also been linked to alkalinity, which is influenced largely by basin geology and physio-chemical weathering (Kwak and Waters 1997). Stream susceptibility to acidic deposition and the resultant biotic community are strongly correlated with stream alkalinity and watershed buffering capacity (Pinder and Morgan 1995, Welsh and Perry 1997, Baldigo and Lawrence 2000).

Many fish species are habitat specialists (Felley and Hill 1983, Gorman and Karr 1978) and consequently species richness has been linked to heterogeneity of habitat types (Gorman and Karr 1978, Schlosser 1987, Guegan et al. 1998, Rathert et al. 1999, Grenouillet et al. 2004). Although habitat generalists are often ubiquitously distributed (Pirhalla 2004), their relative abundance may be dictated by competitive interactions and predation (Mendelson 1975, Taniguchi and Nakano 2000, Galbreath et al. 2001).

Detecting habitat influences at scales relative to fish presence or relative abundance is difficult (Fausch et al. 2002). Furthermore, inferences from data measured at one spatial scale may be irrelevant at other spatial scales (Levin 1992). Habitat collected at intermediate spatial scales may be most applicable to predicting species presence/absence and relative abundance (Vadas and Orth 2000, Peterson and Rabeni 2001, Fausch et al. 2002).

Many lotic communities are structured along gradients of hydrologic variability and predictable species assemblages occur along this disturbance gradient (Kinsolving and Bain 1993, Poff and Allan 1995). Fish response to hydrologic variability is commonly species-specific as well as life-stage specific (Schlosser 1985, Aadland 1993).

Furthermore, fish assemblage structure may be linked to both extreme flood and drought events (Poff and Allan 1995, Grossman et al. 1998).

Our understanding of how local and regional factors interact across spatial scales to influence fish assemblages is not currently well developed (Fausch et al. 2002). Consequently, watershed scale consequences of extensive impairment are poorly understood. However, we are developing an improved understanding of the role of dispersal (Riley et al. 1992, Gowan et al. 1994, Smithson and Johnston 1999) and stream network geometry (Gorman 1986, Osborne and Wiley 1992, Schaeffer and Kerfoot 2004) on stream fish assemblage structure. Dispersal and stream network geometry are central to the emerging riverscape view in stream fish ecology (Ward 1989, Fausch et al. 2002).

A changing view of the role of dispersal in fish population and assemblage regulation is vital in the dynamic landscape model (Schlosser 1995a, 1995b, Schlosser and Angermeier 1995). Until recently, resident stream fishes were viewed as relatively immobile (Gerking 1953). However recent examinations have discovered that a segment of most populations are mobile and dispersal plays a critical role in the maintenance of stream fish populations (Gowan et al. 1994, Smithson and Johnston 1999). The role of dispersal is dependent upon physical and chemical conditions as well as stream position within the drainage network.

Stream position within the drainage network, using Tonn et al.'s (1990) filter analogy, can be viewed as an additive filter where stream reaches at the margins of a species' distribution core may gain species by providing refuge, foraging, or reproductive habitats that supplement their core habitat areas (Dunning et al. 1992, Schlosser 1995, Schlosser and Angermeier 1995). Areas of high fish diversity such as large streams and

rivers, reservoirs, or lakes may act as species sources for small streams nearby, therefore leading to unusually high species diversity in small streams (Gorman 1986, Osborne and Wiley 1992, Schaeffer and Kerfoot 2004).

Individual fish species are expected to differ in their response to local and regional influences. An understanding of the life history characteristics of fishes within the regional species pool is vital to understanding and predicting the distribution of individual fish species relative to physical, chemical, spatial, and biological characteristics of their aquatic environments (Tonn et al. 1990, Schlosser and Angermeier 1995). Individual species' response to the spatial organization of suitable patches is dependent upon dispersal ability, biotic interactions, and demographic processes (Schlosser 1995a, Schlosser 1995b, Schlosser and Angermeier 1995).

Local impairment can produce regional consequences in the following ways. First, disruption of connectivity within the stream network is expected to have regional consequences through several potential pathways. Barriers to dispersal such as culverts (Warren and Pardew 1998), dams (Winston et al. 1991, Schmetterling 2003), or severe chemical degradation (Atland and Barlaup 1996, Woodward et al. 1997) may isolate populations. Isolation of populations removes potential complementary habitats (Schlosser 1995a) that may be important for long-term persistence of the population. Isolation has also been shown to lead to local extirpations (Dunham et al. 1997, Donalson and Nisbet 1999). Removal of access to supplementary habitats (Schlosser 1995a) may reduce population demographics and long-term viability (Lee and Rieman 1997). Isolated populations tend to be more variable (House 1995) and population variability has been linked to population extirpation (Taylor and Warren 2001). Additionally, extirpated

local populations are less likely to become repopulated in isolated stream reaches (Ensign et al. 1997). Highly mobile fishes or fishes that migrate to spawn are most likely to be impacted by barriers to dispersal (Thurow et al. 1997, Warren and Pardew 1998). Labbe and Fausch (2000) and Scheurer et al. (2003) both present evidence of the importance of watershed connectivity at a number of spatial scales and the importance of intermediate or segment scale (1 – 100 km; Fausch et al. 2002).

Second, activities that reduce demographic processes within the core habitat area (Stacey and Taper 1992, Schlosser and Angermeier 1995) and reduce the potential source of colonizing species or individuals may produce regional effects. Degradation to a population source (Pulliam 1988) is most likely to have regional consequences. Impact to a source population is expected to impact associated sink habitats (Pulliam 1988, Labbe and Fausch 2000). Local extirpations and decreased demographic rates are likely in heavily and moderately degraded stream reaches (Pulliam 1988, Schlosser and Angermeier 1995, Labbe and Fausch 2000). These effects may be transferred upstream (Fausch et al. 2002), resulting in an alteration of species richness or assemblage structure. Measured reductions in species richness, loss of expected species, or quantifiable alteration of assemblage structure are all evidence of regional impairment.

I expect that the interaction between watershed connectivity and local degradation is likely to have regional effects. The effects of direct anthropogenic effects on fishes are well documented (e.g. Karr 1981, Woodward et al. 1997, Wang et al. 2000). Less obvious are the indirect effects that degraded water chemistry may have on watershed connectivity and fish assemblages. Studies involving the upstream effects of dam, culverts, or other barriers (Winston et al. 1991, Warren and Pardew 1998) and their

removal (Kanehl et al. 1997, Bednarek 2001) provide the best available information on the importance of watershed connectivity. However, to my knowledge no published studies have examined the potential watershed scale effects of degraded water quality on stream fish communities.

Acid mine drainage (AMD) impacted watersheds provide an ideal system to assess the potential for regional impairment as a result of extensive local impairment. Acid mine drainage impacted systems are relatively common in the central Appalachian Mountain region. Within the mid-Atlantic and Southeastern United States, it is estimated that 4590 km of streams are severely impacted and another 5780 km were strongly impacted, but not acidic (Herlihy et al. 1990). In the Northern Appalachian subregion of the National Stream Survey (NSS), approximately 10% of stream reaches were acidic due to AMD (Herlihy et al. 1990). Acid mine drainage occurs when pyrite-rich bedrock and residual coal from mining activities reacts with air and water forming a strong sulfuric acid (Herlihy et al. 1990). Aside from low pH, AMD-laden water generally has high concentrations of heavy metals and sulfates and acid neutralizing capacity has been exhausted (Herlihy et al. 1990, WVDEP 1996, Williams et al. 1999). In addition, throughout much of the Appalachian Region, this degradation has been part of the landscape over a long period of time. Much of the current degradation occurred from mines that operated before the Surface Mining Control and Reclamation Act of 1977 was initiated. However, many of these same mines are currently discharging untreated AMD.

Acid mine drainage may affect fish assemblages through both direct and indirect interactions. Fish assemblages in AMD-impacted reaches are directly impacted

predominantly through heavy metal toxicity (Barry et al. 2000, Lydersen et al. 2002), disruption of ion regulation (Beaumont et al. 2000), and reduction of benthic macroinvertebrate density (Scullion and Edwards 1980). My earlier investigations (Freund Chapter 1) determined both macroinvertebrate and fish bioassessment scores were greatly reduced in areas directly impacted by AMD. Potential indirect effects of AMD on fish communities include: reduced dispersal through avoidance behavior of fishes (Woodward et al. 1997), a reduced source of potential colonizing individuals (Gore and Milner 1990), and creation of temporal chemical barriers during which dispersal is limited to periods of relatively good water quality, analogous to Schlosser's (1995a) semi-permeable barrier; or in the most heavily impacted streams, AMD may act as a true barrier to dispersal, analogous to a dam (Winston et al. 1991). Acid mine drainage often provides a severe and extensive degradation of habitat quality (Herlihy et al. 1990, Williams et al. 1999). Consequently, AMD has the potential to produce a significant regional impact.

Given the potential for regional consequences as a result of extensive reach scale water quality impairment, I addressed the following objectives.

1. Identify the physical, chemical, and spatial features that influence fish assemblages and species distributions in an unimpaired sub-basin of the Cheat River watershed.
2. Use this information to quantify the local and regional effects of mining on fish communities in an intensively mined sub-basin of the Cheat River watershed.

Methods

Study Area

The Cheat River is part of the upper Ohio River basin and is formed by the confluence of Shavers Fork and Black Fork Rivers in Parsons, WV. From its origin, the Cheat River flows north approximately 135 km before entering the Monongahela River near Point Marion, PA. The Cheat River drains an area of approximately 3700 km², almost entirely within West Virginia. Cheat Lake, a large impoundment of the Cheat River, is at the Pennsylvania-West Virginia border and was the downstream boundary for this study. Much like watersheds throughout the upper Ohio River basin, the Cheat River has a relatively low diversity fish community, supports a few rare species and no endemic species (Stauffer et al. 1995). Tributaries to the Cheat River tend to be dendritic and headwater streams are generally cold, whereas many larger second and third order streams support warmwater fish assemblages.

For the purposes of this research, the Cheat River basin was divided into upper and lower basins. The lower basin (i.e., the northern portion of the watershed) encompasses the area downstream of Pringle Run, the first major source of untreated mine drainage. Many streams within the lower basin are directly impacted by AMD. However, numerous unimpacted streams remain and provide an ideal opportunity to quantify indirect effects that may result from extensive AMD impairment. Several lower basin sites, based on water chemistry and macroinvertebrate bioassessment scores, rate among the best in the Cheat River watershed (WVDEP 1996). Streams within the upper basin are unimpacted by AMD, but experience many of the same anthropogenic disturbances as those in the lower basin (acid precipitation, development, agriculture, and silviculture; WVDEP 1996). The upper basin is minimally impacted (WVDEP 1996) and

provides the ideal reference watershed to test the local and regional consequences of AMD in the lower basin.

Geology within the Cheat River basin consists predominantly of Mauch Chunk shale and Pottsville Sandstone (Cardwell et al. 1968). Greenbrier limestone is limited in its distribution but is locally important in both the upper and lower basin as it provides relatively high amounts of alkalinity (Welsh and Perry 1997). Mauch Chunk produces a limited amount of alkalinity whereas Pottsville sandstone produces little to no alkalinity (Welsh and Perry 1997) and is commonly associated with high sulfur coal seams (WVDEP 1996).

Sampling Design

Twenty-seven upper basin and 37 lower basin sites were sampled within the Cheat River watershed (Figure 1). All sites were tributaries to the Cheat River and were selected based upon their location in the watershed (upper or lower basin), proximity to larger streams (spatial position), basin area, and lithology. Based on previous research (WVDEP 1996), water quality and stream condition, and macroinvertebrate assemblages, was known. Within the lower basin, sites were selected to represent the range of AMD impairment from completely unimpaired to severely impaired.

Sites had a minimum drainage area of 3 km² and no predetermined maximum drainage area although wadeable sites were 3rd Strahler order or smaller. McCormick et al. (2001) found that streams with basin areas smaller than 3 km² were often devoid of fish in the Mid-Atlantic Region. I ensured that both the smallest (3.2 km²) and the largest (141.0 km²) sites that were sampled were located in the upper basin to ensure that models developed in the upper basin and then applied to the lower basin did not extrapolate

based on basin area. Within this range of 1st – 3rd order streams, sites were selected to best represent the range of geologic variability. Site selection was weighted by spatial position following Osborne and Wiley (1992) to incorporate the influence of spatial position, a potential regional effect, on stream fish assemblages. Sites over the range of basin area were selected so that we sampled streams at various distances from a species pool, defined as 4th order or larger for this study (e.g. Big Sandy Creek and Cheat River). I did not sample sites in proportion to their availability, rather site selection covered the range of basin area variability so that intermediate and large basin areas were sampled disproportionately to their availability.

Physical Habitat Data

Rapid visual habitat assessment (RVHA; Barbour et al. 1999) data were collected at all sites coinciding with fish sampling following protocols in Barbour et al. (1999) for high gradient streams. Total RVHA score is a composite of 10 metrics: epifaunal substrate/available cover, embeddedness, velocity/depth regime, sediment deposition, channel flow status, channel alteration, frequency of riffles (or bends), bank stability, vegetative protection, and riparian vegetative zone width. The highest possible total score is 200 (100%) while scores greater than 160 (80%) were considered optimal, 110-159 (55-79.5%) scores are suboptimal, scores of 60-109 (30-54.5%) are with marginal range, and scores less than 60 (30%) indicate poor habitat conditions. Visual habitat surveys were completed by either a single trained observer or two trained observers that arrived at a consensus. Surveys were completed after fish sampling to ensure that observers had an opportunity to view the entire stream.

Water Chemistry Sampling

Water quality data were collected in April 2003, sampling was conducted to coincide with the poorest water quality in moderately impaired sites (Petty and Barker 2004) and with WVDEP watershed assessment sampling protocols (WVDEP 1996). At each sample location discharge was determined by multiplying stream width, average depth, and average current velocity. Temperature (C), pH, specific conductance ($\mu\text{S}/\text{cm}^3$), dissolved oxygen (mg/l) and total dissolved solids (mg/l) were measured with a YSI 650 and a 600 XL sonde (Yellow Springs, Ohio). A 500 ml sample was filtered with a Nalgene polysulfone filter holder and receiver. Each filtered sample was passed through a mixed cellulose ester membrane 0.45 μm pore size disk. Filtered samples were treated with 5 ml 1:1 nitric acid to bring the pH below 2.0 to prevent dissolved metals from falling from solution. Filtered samples were used for analysis of total dissolved aluminum (mg/l), iron (mg/l), manganese (mg/l), nickel (mg/l), cadmium (mg/l), chromium (mg/l), and calcium and total hardness (mg/l). A 1 liter grab sample was collected for analysis of alkalinity (mg/l CaCO_3 eq.), acidity (mg/l CaCO_3 eq.), and sulfate (mg/l). Unfiltered samples were kept on ice after collection and stored in the laboratory at 4°C until analysis could be completed.

All laboratory analyses were conducted at Black Rocks Test Lab in Morgantown, WV, using procedures from the 18th edition of Standard Methods for the Examination of Water and Wastewater (Clesceri et al. 1992). Acidity and alkalinity as calcium carbonate (CaCO_3) were determined using the titration methods (methods 2310 and 2302B, respectively). Sulfate was determined using the turbidimetric method (method 426C). Iron, manganese, nickel, cadmium, and chromium were analyzed with AAS (atomic

absorption spectrophotometer) using method 3111B. Aluminum was analyzed using an AAS, using method 3111D. Hardness as CaCO_3 was measured using an AAS and calculations from method 3111B.

A subset of sites were sampled regularly over a single year to determine the effectiveness of a single sample in quantifying relative water quality. Analysis of this data set by Petty and Barker (2004) suggests that the early spring sample used in this study can be used to effectively identify the worst conditions experienced by moderately AMD-impacted streams. In addition to mining effects, acidic conditions related to acid deposition are most common during spring (Driscoll et al. 2001). Petty and Barker (2004) provide a detailed summary of data collection and an analysis of water chemistry parameters and their variability within the Cheat River basin.

Stream Size and Spatial Position

Variables pertaining to stream size and spatial position were obtained from GIS coverages from West Virginia University Natural Resource Analysis Center (WVU NRAC). Stream size variables were: basin area (km^2), Strahler and Shreve orders, and downstream link. Spatial position variables included link order difference, 2nd, 3rd, and 4th link order differences, and distance from a large river source (m). Coverages were produced from 1:24,000 maps and cell size was 30m x 30m. Analysis and data collection were done using Watershed Characterization and Modeling Systems Version 2.8 for Arc View 3.2 (Strager 2002) and ArcMap 8.1.

Fish Data Collection

Fish data were collected from 15 June through 15 October during 2002 and 2003. Fish sampling protocols were modified slightly from EPA-EMAP protocols outlined by

McCormick et al. (2001). Stream reaches were 40 times the mean stream width (MSW) or a minimum of 150 m and a maximum of 300 m in length (Lyons 1992, Yoder and Smith 1999). Sampling greater distance typically do not yield significant improvements in IBI score or species richness (Ohio EPA 1989). Protocols specified that shocking effort was not to exceed 1800 seconds; measured as the amount of time in seconds that electrical current is applied to the stream (Ohio EPA 1989). This time rule was only evoked for only one site in which 200m of the 300m were sampled and over 3,000 individual fish were collected. Sites generally included at least two riffle-pool sequences and started at the base of a riffle. When possible, attempts were made to not include channelized or intensively human altered sections.

Electrofishing consisted of a thorough single pass, recommended for large-scale geographic studies (Meador et al. 2003) and in most bioassessment protocols (Barbour et al. 1999, Yoder and Smith 1999, McCormick 2001). A single electrofishing pass may underestimate species richness and testing efficiency using multiple passes is suggested (Meador et al. 2003). In 2001, preliminary research was conducted to assess electrofishing efficiency. Three streams were sampled using three passes and a fourth stream was sampled with two electrofishing passes. Only one species that was not captured in the first pass was captured in subsequent passes. Over the four sites, 61% of individuals were captured in the first pass and 95% of species were captured in the first pass.

Sampling protocols varied slightly depending upon stream size and structural diversity. Block nets were set at the upstream and downstream reaches when width, discharge, or materials that were likely to cause block nets to fail were not limiting. In

small or structurally simple streams, a single backpack electrofisher was used, whereas in larger streams two or three electrofishers were used in concert. All electrofishing efforts were supported by a portable block net. Electrofishing operators began shocking upstream from the portable blocknet for 10 m or until coming to a natural barrier (riffle, depth change, etc.), whichever distance was shorter. At the end of the upstream shocking efforts, the operators worked back to the portable block net dislodging stunned fishes that had become trapped in the substrate or instream structure. Depending upon the stream size, one to six workers with nets collected stunned fishes and assisted in dislodging and collecting stunned fishes. Upon the crew returning to the portable block net, the net was lifted from the water and fishes were collected from the net. Fishes were stored in 20 L buckets until they could be placed in flow-through livewells placed in the stream. Fish were processed and recorded separately for each 50 m section to minimize the time that they were confined to the instream livewells.

After each 50 m section was sampled, fish were identified to species and returned to the stream. In a subset of streams, 40 individuals of each species were randomly selected to be weighed and measured to determine a length-weight relationship for each stream and for the Cheat River watershed. Voucher specimens were collected and preserved in a 10% buffered formalin solution in 2002 or 95% ethyl alcohol in 2003 to allow for future genetic research. A subset of fishes of each species were vouchered for later identification verification. Individual fishes that could not be identified in the field were vouchered to be compared to reference collections of West Virginia University and the West Virginia Division of Natural Resources. Identifications were verified by West Virginia University and the West Virginia Division of Natural Resources personnel.

Statistical Analyses

Using SAS 8.1, continuous variables were natural log transformed, and proportional data (RVHA components and total score) were arcsine square root transformed to improve normality (Zar 1999). To aid in testing my objectives, I classified sites based on basin (i.e., upper or lower), distance from a mainstem river (i.e., near or far), and AMD-impairment (i.e., non-impaired, moderately impaired, or severely impaired). Cheat River tributaries upstream of Pringle Run comprised the upper basin. All sites downstream of Pringle Run, the first source of AMD-impairment, were classified as lower basin sites. Categorization of AMD-impairment was based on water chemistry data and known history of mining within the basin. For lower basin unimpaired and moderately impaired sites, I categorized sites based on their distance from a mainstem river (Osborne and Wiley 1992). To match my original study design, one km was the cutoff between “near” and “far” sites.

Principal components analysis (PCA) was used to explore covariance among variables in the basin area and spatial position, water chemistry, and RVHA data sets. Factors with an eigenvalue greater than 1.0 were considered to be significant. Individual factors with an absolute value greater than 0.30 were considered to be significant and those with an absolute value of 0.40 were considered to be more important (McGarigal et al. 2000).

Discriminant function analysis was used to test the ability of lower basin sites to be classified into AMD impairment categories (i.e. non-impaired, moderately impaired, and severely impaired) based on water chemistry PC1 and PC2 factor scores. Severely impacted sites formed a unique group that was easily distinguishable from moderately

impaired sites. Unimpacted sites were similar in water chemistry and PC1 score to upper basin sites. Discriminant function analysis (DFA) was used to test if AMD categories accurately captured the two predominant PCA factor loading scores. Any sites that did not correctly classify and did not have known sources of mining-related discharge were placed into categories predicted by DFA classification/misclassification analysis.

Significant PCA factors from the water chemistry data set were not included in models applied to the lower basin as we had *a priori* knowledge that water chemistry strongly influenced biotic communities in the lower Cheat River watershed (Freund Chapter 1, WVDEP 1996). Additionally, since my objectives were to develop models using the non-AMD impacted upper basin as a reference basin as a test of the direct and indirect effects of water quality impairment in the lower Cheat River basin, measures of water chemistry could not be used to determine expected patterns that would be applied to the lower basin. To test these objectives, water chemistry PCA factors were used to explain observed deviation from expected patterns but were not included in predictive models.

I used multiple regression to build models to predict natural variability in fish species richness in the upper basin as a function of physical (e.g. habitat quality) and spatial (e.g. stream size, spatial position) characteristics. Models constructed in the upper basin were then applied to the lower basin. Observed/expected values were used to detect both local (i.e. reach scale) and regional consequences of water quality impairment from AMD.

Stepwise multiple regression models were developed for the upper basin using variables from the area and spatial position, water chemistry, and RVHA data sets based

on my *a priori* selection criteria. Principal component factor loadings were retained as independent variables in the development of species richness models if they provided additional information regarding covariance in the data sets that could not be accurately explained by individual by a single variable or multiple variables that were highly correlated with significant PCA factor scores. If PCA factors did not meet this criteria, correlation analysis was used to assist with variable selection in order to reduce redundancy among variables entered into the stepwise multiple regression models. Parameters were left in the stepwise model if the p-value was less than 0.15, although the full model was assessed at the $p = 0.05$ level.

The species richness model developed in the upper basin was applied to the lower basin to set expectations for species richness based on habitat, spatial position, and stream size. Relationships between observed and expected species richness were examined using simple linear regression for lower basin sites as well as for AMD-impairment categories in the lower basin. Similarly, relationships among the ratio of observed and expected species richness and basin area and water quality PC1 were tested at the basin and AMD-impairment level scales. A similar, multiple regression, species richness model for the upper basin was developed that included water chemistry PC1 and PC2 in the model building process to determine if water chemistry explains a significant amount of variation in species richness in the lower basin. Similarly, a species richness model for the lower basin was developed including water chemistry PC1 and PC2 to determine how much of the variability in lower basin species richness could be explained by water chemistry measures.

I used DFA to build multivariate models to predict presence/absence of several commonly occurring species in the Cheat River basin. As with the species richness models, these models were developed in the upper basin and were then applied to streams in the lower basin to detect local and regional effects of AMD on fish communities. Predicted presences and absences from the DFA model were compared to observed species distributions to determine classification strength for each species. Discriminant function analysis classification/misclassification analysis was then used to determine the ability of the model to predict species presence in each basin. Correctly predicted presences for each lower basin site were set as common species richness expectations and were compared to observed common species richness in the lower basin. Comparisons were made among the AMD groups to determine how well commonly occurring species could be predicted in unimpaired and moderately impaired lower basin sites.

To assess what factors are likely to be responsible for individual species presence/absence in the lower basin, I developed similar presence/absence models for all lower basin sites and for unimpaired and moderately impaired sites. In developing these models, the same variables as the upper basin were used with the addition of water quality factor scores. Again, following the process from the upper basin, classification/misclassification was run to test the models' fit to the data. Deviation in variable selection or classification strength was expected to indicate large-scale deviation from upper basin patterns.

Finally, I developed a new "core/periphery" approach to describe the distribution of individual fish species as a function of stream size (i.e., basin area). The objective of this approach was to classify each fish species with regard to the extent of its distribution

range (i.e., narrow vs. wide-ranging) and the position of its core range along the stream size continuum (i.e., small stream vs. large river). Again, information from the upper basin was applied to the lower basin and used to detect which species were most susceptible to direct, local AMD impacts and which were susceptible to regional scale losses.

Core habitat area was defined as the range of basin area representing the central 50th percentile of the cumulative relative frequency of individuals captured for each species. Species core area provides a conservative estimate of the range of basin areas over which a species is distributed. The observed cumulative relative frequency distribution was compared to the accumulation of area sampled. The cumulative relative frequency distribution for area sampled represented the expected value for species accumulation under the null hypothesis that each fish species accumulated individuals relative to the area of stream sampled. Kolmogorov-Smirnov goodness of fit for continuous data was used to test for differences between the observed and expected cumulative relative frequency distributions (Zar 1999). A non-significant result indicated that the species accumulated in proportion to the area of stream sampled.

Habitat core areas were used to categorize each species as small stream specialist, intermediate stream specialist, large river specialist and ubiquitous depending upon the extent of the core, the 50th percentile of their cumulative distribution, distribution of fish density to core range, and the relationship to the expected distribution. Ubiquitous species had a cumulative frequency distribution that did not differ from the expectations set by the sampled area. Small stream species accumulated individuals in small streams at a significantly faster rate than expected or by chance alone. Conversely, large river

species accumulated individuals in large streams more quickly than expected or by chance alone. Finally, intermediate specialists accumulated individuals at a lower than expected rate in small and large streams and at a faster than expected rate in intermediate sized streams.

The extent of the core describes the range of basin area over which the species' core habitat area extends. Species with a narrow core extent have a steep slope associate between the 25th and 75th percentiles of cumulative distribution and are generally limited to a narrow range of basin area. Species possessing a wide core extent have a relatively low to moderate slope between the 25th and 75th percentiles. These species have a core area that extends over a wider range in basin area. Periphery areas were defined as the range of basin area that were not included within the core range of each species. For many species, periphery areas are expected to be maintained through their interaction with adjacent core areas. Additionally, occupancy rates of periphery areas are expected to be more temporally variable.

Core area and periphery percent occupancy were determined for each species over all sites in the Cheat River basin. Occupancy rates were compared between the upper and lower basins and differences in percent occupancy was determined. Additionally, core and periphery site occupancy was determined for unimpacted sites in the lower basin and compared to upper basin occupancy rates as a measure of regional impairment.

Results

Utility of PCA in Partitioning Covariation among Variables

Principal component analysis was used to describe covariance in spatial position and stream size, RVHA data, and water chemistry data sets. Ordination of water

chemistry data suggests sites followed an AMD (PC1) and a potential acid neutralizing gradient (PC2; Table 1, Figure 2). Impairment as a result of AMD in the lower basin, depicted by PC1, was distributed over the range of stream sizes that were sampled (Figure 3) and was not correlated with basin area ($R^2 < 0.01$, $df = 35$, $p = 0.7555$). The third and fourth significant axes had lower eigenvalues than the first two axes, explained little of the variation in water chemistry, and were difficult to interpret. The first principal component (PC1) of the RVHA data set was strongly correlated with total RVHA score ($R^2 = 0.96$, $df = 52$, $p < 0.0001$) and did not provide information that total RVHA score did not provide. Principal component analysis of the spatial position and stream size data set produced two significant axes, neither of which yielded any information regarding covariance among stream size and spatial position variables.

Assessing AMD-Impairment Categories

Variable selection for inclusion in species richness and individual species presence DFA models relied upon information provided by PCA and correlation analysis of the three data sets. From the RVHA data set, total score was entered into future models based on the strong correlation between the first PC axis and total RVHA score. From the spatial position and stream size data set basin area, Shreve and Strahler orders, downstream link (D-link), and downstream link difference were selected based on PCA factors scores and examination of the correlation matrix. These variables were relatively weakly correlated. Analysis of the correlation matrix for the data set determined that other measures of stream size and spatial position variables were redundant with the variables selected above.

Discriminant function classification/misclassification analysis of AMD-impairment categories of none, moderate, and severe yielded only two misclassifications. One lower basin site originally labeled as non-impaired was classified with the moderately impaired group. This site was moved into the moderately impaired group for all future analyses. A second lower basin site was originally classified as severely impacted but DFA grouped it with the moderately impaired sites. However, because no fish were present when the stream was sampled and a known history of AMD treatment, I retained the site in the severely impaired group.

General Trends in Fish Assemblages

Differences among fish species richness and tolerance categories (McCormick et al. 2001) were evident between the two basins (Table 2). Forty-one fish species were sampled from 66 sites within the Cheat River basin. Thirty species were sampled in the upper basin and 37 fish species were sampled from the lower basin. The difference in basin-wide species richness was due to large river cyprinid species and a few lentic species associated with impoundments in the lower basin, Cheat Lake in particular. These same fishes were not collected in the upper basin. The average site species richness for the upper basin was 11.0 with a coefficient of variation of 9.0%. In the lower basin, average fish species richness was 5.7 with a coefficient of variation of 19.7%. Deviation in average species richness between the two basins was linked to ten lower basin sites that were fishless. Within the upper basin, approximately 20% of individuals in each site were classified as tolerant species and 7% were classified as intolerant according to criteria from McCormick et al. (2001; Table 3). Lower basin sites had a higher proportion of tolerant species than the upper basin (t test; df = 61, p-value = 0.005)

In addition to patterns in species richness and tolerance levels, differences in the frequency of occurrence of individual species were also evident between the two basins. For example, mottled sculpin (*Cottus bairdi*) were collected in every upper basin site whereas in the lower basin they were collected in 51% of all sites and 70% of sites that were not fishless. Creek chubs (*Semotilus atromaculatus*) were found in an equivalent number of non-fishless lower basin sites and upper basin sites. White sucker (*Catostomus commersoni*) and rainbow trout (*Onchorhynchus mykiss*) were the only species that were found in a greater proportion of non-fishless lower basin sites in comparison with upper basin sites. Aside from these three species, all other species occurred in a greater proportion of upper basin sites (Table 3).

Species Richness Models

Two variables retained by the stepwise multiple regression explained 79% of the variability in species richness in the upper Cheat River basin (Figure 4). Basin area (km², log transformed) explained most of the variability in species richness (partial $R^2 = 0.75$). Link order difference explained 4% of variation in species richness not explained by basin area.

Fitting the upper basin model to the lower basin produced three important results. First, the overall fit of the model to the lower basin data was poor ($R^2 = 0.34$ for lower basin vs. $R^2 = 0.79$ for upper basin). Two, the general tendency was for the lower basin sites to possess lower than expected species richness, regardless of local water quality. Thirty-two of the 37 lower basin sites possessed lower than expected species richness. This included all four severely impacted sites, 15 of 18 moderately impaired, and 10 of 12 non-impacted sites. Three, only larger streams (i.e. streams with a higher expected

richness) in the lower basin possessed species richness values that exceeded expectations. In fact, several large moderately impaired streams possessed species richness values that were near to or exceeded expectations based on the upper basin model (Figure 5).

Further examination of the relationship between observed and predicted species richness yielded several important observations. First, the ratio of observed species richness and expected species richness was not a function of basin area in either the upper ($R^2 < 0.01$, $df = 21$, $p = 0.87$) or lower ($R^2 = 0.09$, $df = 35$, $p = 0.0840$) basin. Similarly, neither unimpaired sites lower basin ($R^2 = 0.20$, $df = 12$, $p = 0.13$) and moderately impaired sites ($R^2 = 0.14$, $df = 18$, $p = 0.1087$) were not correlated with basin area. Lower basin observed/expected ratio also showed weak correlation with water quality PC1 in the lower basin (Figure 6; $R^2 = 0.19$, $df = 35$, $p = 0.0080$) and was non-significantly correlated when severely-impacted sites were removed from analysis ($R^2 = 0.10$, $df = 31$, $p = 0.0780$).

A species richness model for the lower Cheat River basin developed including water chemistry PCA factors yielded a total model R^2 of 0.54. Shreve order explained a majority of the variability (31%) while water quality PC1 (10%) and D-link (13%) also contributed significantly to the lower basin model.

Species Presence/Absence Models

Development of discriminant function models to explain fish species presence/absence in the upper basin included the same variables used to develop the species richness models. These presence/absence models for the upper basin generally had strong classification strength. However, these same models performed relatively poorly when applied to the lower basin (Table 4). Species presences and absences in the

upper basin were typically predicted with greater than 80% accuracy, whereas prediction accuracy for the lower basin was highly variable and much lower overall. Basin area or related measures of stream size were retained in the development of stepwise discriminant function models for all individual species (Table 5). For some species, presence/absence was also linked to spatial (i.e., distance from mainstem) or physical habitat (i.e., RVHA score) site characteristics. Discriminant models could not be developed for several species (e.g., mottled sculpin; blacknose dace *Rhynchichthys atratulus*), and longnose dace *Rhynchichthys cataractae*) that occurred in greater than 90% of upper basin sites. These species were expected to occur in all lower basin sites based on their distribution in the upper basin.

When individual presence/absence models were compared with the distribution of fishes in the lower basin, several important trends were discovered. In comparison with the upper basin, rates of correctly predicted presences by the models were poor (Table 6). Only rock bass (*Ambloplites rupestris*), mottled sculpin, blacknose dace (*Rhynchichthys atratulus*), creek chub, and brook trout (*Salvelinus fontinalis*) were collected in 50% or greater of the sites in which they were predicted to be present in the lower basin. Differences between correctly predicted presences in unimpaired and moderately impaired lower basin sites made it difficult to determine general trends among all species. However, examination of individual species leads to some potentially important patterns. Mottled sculpin, greenside darter (*Etheostoma blennioides*), smallmouth bass (*Micropterus dolomieu*), brook trout, and longnose dace presences were predicted with more accuracy in unimpaired sites compared to moderately impaired sites. Only rock

bass, white sucker, and *Notropis spp.* were more often correctly predicted to be present in moderately impaired sites than in unimpaired sites.

Similar to the results from the species richness model developed for all species, observed richness of commonly occurring species, as predicted by the presence/absence models, fell well below expectations (Figure 7). Observed commonly occurring species were significantly correlated with expected common species richness for moderately impaired sites ($R^2 = 0.34$, $df = 13$, $p = 0.0296$) but not unimpaired sites ($R^2 = 0.03$, $df = 12$, $p = 0.6323$). The number of commonly occurring species in unimpaired sites was highly variable. However, richness in unimpaired lower basin sites was generally much reduced from expectations. Again, similar to the species richness model for all species, two moderately impaired and one unimpaired site had greater common species richness than predicted.

Similar presence/absence models were developed for the lower basin to examine how patterns of presence/absence in the lower basin differ from the upper basin. Stepwise discriminant function model development for lower basin sites shows that stream size related variables (i.e., basin area, stream order, D-link) explained much of the variation for many species. A measure of stream size was included in the final model for all species with the exception of smallmouth bass (Table 7). Water quality factor scores were also commonly retained in models but physical habitat (i.e., RVHA score) and measures of spatial position (i.e., distance from mainstem) were rarely retained. Similar to the presence/absence models developed for the upper basin, classification strength of both presences and absences when compared to observed values were generally strong (Table 8).

When similar presence/absence models were developed for unimpaired and moderately impaired sites separately, retention of variables in stepwise discriminant models varied greatly. Model development in unimpaired sites (Table 9) often failed to yield a significant model. Additionally, retention of variables differed greatly from those included in upper or lower basin models and from models developed in moderately impaired sites (Table 10). Models for moderately impaired sites more often included measures of stream size, similar to the results of model development for upper and lower basin sites. Presence/absence models for unimpaired sites always correctly predicted species presences (Table 11). Models developed for moderately impaired lower basin sites also performed well.

Species Core/Periphery Models

Core habitat area provided useful measures of species distribution (Table 12). The 25th, 50th, and 75th percentiles of each species' relative cumulative proportion of total individuals provided measures of distribution of each individual species as a function of basin area. Their core range, the central 50th percentile of their distribution, is a measure of the basin area range over which the species are most likely to occur. Occupancy rates for sites within core habitat range were generally high in the upper basin, averaging 86%, whereas occupancy rates for periphery sites was 51%. Occupancy rates in lower basin sites were generally much lower (Table 13).

Comparison of core and periphery occupancy rates between the upper and lower basins and in non-AMD impaired sites provided information about both regional and local effects of water quality degradation in the Cheat River basin. Core and periphery occupancy rates in the lower basin were 48% and 32% lower, respectively, than they

were in the upper basin (Table 13). Additionally, occupancy rates of core and periphery sites in the upper basin were higher for all species with the exception of white sucker, which occupied 18% more of the sites within their core range in the lower basin compared with the upper basin. Core and periphery occupancy rates in non-AMD impacted lower basin sites, 68% and 19% respectively, differed from core (86%) and periphery (51%) occupancy rates in the upper basin (Table 14). This difference was most pronounced in periphery sites for most species.

Four important patterns in core habitat types were evident (Table 15). First, a subset of species was ubiquitous over the range of stream sizes we sampled aside from the smallest and coldest of streams. Several common ubiquitous core species were mottled sculpin (Figure 8a), blacknose dace (Figure 8b), longnose dace (Figure 8c), and creek chub (Figure 8d). These species also comprise four of the most widely distributed species in the upper basin.

Second, brook trout were the only species whose density was strongly negatively correlated with basin area and displayed a small stream core area (Figure 8e). Despite the smallest core range, brook trout were found in both the largest and smallest sample sites in the upper basin and were the fourth most widely distributed species in the upper basin and the fifth most widely distributed in the Cheat River watershed. Although brook trout were present in larger basin area sites, they were found in relatively low densities in larger streams, perhaps suggesting a high rate of dispersal by this species (Figure 8e).

Third, the largest subset of species had their greatest densities and accumulated most rapidly in moderate to large streams. These intermediate stream core species are represented by rock bass (Figure 8f), white suckers (Figure 8g), Hybrid minnows (Figure

8h), northern hogsuckers (Figure 8i), green sunfish (*Lepomis cyanellus*; Figure 8j), and river chubs (*Nocomis micropogon*; Figure 8k). While these species showed a wide range of core ranges, they generally showed the steepest slope for accumulation in intermediate sized streams.

Fourth, large river core species included: central stoneroller (*Camptostoma anomalum*, Figure 8l), greensider darter (Figure 8m), fantail darter (*E. flabellare*; Figure 8n), smallmouth bass (Figure 8o), and combined Notropis species (*N. rubellus* and *N. volucellus*; Figure 8p). The large river core group also likely included several species of catostomids, centrarchids, cyprinids, ictalurids, and percids that were generally collected in larger streams or small streams connected to the Cheat River or Big Sandy Creek but were not collected in a sufficient number to develop a core area model. Examples of rarely collected large core species are black redhorse (*Moxostoma duquesnei*), striped shiners (*Luxilus chrysocephalus*), margined madtom (*Noturus insignis*), and variegate (*Etheostoma variatum*) and banded (*E. zonale*) darters. Rotenone data from the West Virginia Division of Natural Resources for Cheat River sites sampled in 1999 show many fish species that were rarely collected in this study were among the most commonly collected species in the Cheat River (WVDNR unpublished data).

Discussion

Patterns in species richness and individual species presence/absence in the upper basin differed greatly from patterns observed in the AMD-impacted lower basin. The model developed to explain species richness in the upper basin performed poorly when applied to the lower basin. Species occupancy of core and periphery areas was generally much lower in lower basin sites, and occupancy of sites where fish species were

predicted to be present based on the discriminant function models were greatly reduced in the lower basin. Examination of deviations from the upper basin patterns within lower basin sites detected local impairment in severely and moderately AMD-impaired sites. Additionally, regional impairment is evident through the examination of deviation from upper basin patterns within non-AMD impaired lower basin sites.

Upper Cheat River Basin as a Reference Watershed

The upper Cheat River basin is relatively unimpaired and many sites I sampled drained lands within the Monongahela National Forest. Although slight geologic differences exist between the upper and lower basins, much less perturbation has occurred in the upper basin. Development, limited agriculture, and acid deposition are the greatest sources of degradation. In comparison to the lower basin, few roads, development, and agriculture is present in the upper basin and the effects of acid deposition vary with drainage geology (Welsh and Perry 1997). Although truly unimpacted reference watersheds are rare, the upper Cheat basin is minimally impacted and based on similarities with the lower basin, provides the best possible reference watershed by which local and regional impairments in the lower basin can be compared.

Basin area explained much of the variability in species richness in the upper Cheat river watershed. The species-area relationship in the unimpacted upper basin performed similarly to models developed for watersheds in Wisconsin (Newall and Magnuson 1999), New York (Sheldon 1968), and in Angermeier and Schlosser's (1989) research in Minnesota, Illinois, and Panama. Basin area was also important in the development of core-periphery and presence/absence models for individual species. Basin area is the overriding factor determining species richness in most spatial models derived from the

MacArthur-Wilson model species richness developed for oceanic islands (MacArthur and Wilson 1967). Basin area is generally highly correlated with individual fish species distribution and species richness (Sheldon 1968, Angermeier and Schlosser 1989). Basin area is correlated with physical, chemical, and biological factors that have been shown to structure fish assemblages (Vannote et al. 1980, Angermeier and Schlosser 1989). Within the upper basin, the presence and densities of individual fish species were often strongly correlated with basin area.

Following the Theory of Island Biogeography (MacArthur and Wilson 1967), spatial position provided additional predictive power to the species richness model. The addition of a spatial term slightly improved the model, suggesting that proximity to a larger potential species pool can have an additive affect on local species richness (Osborne and Wiley 1992). The species richness model for the upper basin included basin area and link order difference, which is a measure of the relative size difference between the sampled stream reach and the downstream reach.

Similar to findings from other watersheds, the influence of spatial position in the Cheat River basin is highly variable (Grenouillet et al. 2004). Spatial patterns that exist are difficult to elicit due to the limited number of spatial position and impairment level combinations present within the upper basin and underlying natural variability. No specific mechanisms were evident that explained the conditions under which spatial position influenced local species richness. However, several sites, most notably small streams near the Cheat River and within the Horseshoe Run subwatershed, displayed clear evidence of species additions due to spatial position (Osborne and Wiley 1992).

This is supported by the capture of large and intermediate core species in small basin area stream reaches near the Cheat River or larger tributaries to the Cheat River.

Species richness models suggest that species richness in the upper Cheat River watershed are driven by physical and biological gradients associated with basin area (Angermeier and Schlosser 1989), but variability in water chemistry and spatial position add significantly to the basic species-area model. Development of a similar species richness model that included the first two PCA factors in the model selection process allowed the influence of water chemistry to be assessed in the upper basin. Inclusion of water chemistry variables marginally improved the model in the upper basin. The first principal component (PC1) was included in the alternative model despite the low variability in PC1 within upper basin sites. While PC1 was viewed as being sensitive to AMD, metals and sulfates associated with acid precipitation are likely responsible for much of the variation in PC1 within the upper basin. Susceptibility of stream reaches to acid precipitation in the Cheat River watershed is linked to geology and may have been responsible for much of the variability in species richness linked to PC1 (Welsh and Perry 1997). However, water quality, due to the narrow range of variability in the upper basin, had little relative influence in determining species richness.

The core-periphery models established for the Cheat River basin provided a meaningful range over which the species is expected to be securely present and impairment to the core area is likely to be reflected within the periphery. This method attempts to incorporate theoretical spatial models into relatively simple measure of a species central tendency and dispersion. This view provides a measure of stream sizes over which the species is distributed. A narrow core basin area extent suggests a fish

species that has a narrow range of conditions under which it is likely to be persistent. Whereas a wide core range indicates a species that is widely distributed and persistent over a wide range of stream sizes. Impacts to the core range of species with a narrow core range are expected to be potentially more damaging at a regional scale.

The core/periphery view is best viewed as an analogy to the geographic range of individual species but at a smaller spatial scale. Most species occupy a range in which the center of their distribution is generally persistent (Brown et al. 1996, Hayes et al. 1996, Gaston et al. 2000). At the edge of species ranges, suitability is generally reduced and persistence is more tenuous. Likewise, demographic rates near the edge of a species' distribution may be quite different from demographic rates near the center of their distribution (Flebbe 1994, Hayes et al. 1996, Post et al. 1998, Dunham et al. 2003). That the distribution of fish species and their demographic rates are linked to abiotic gradients such as temperature (Flebbe 1994, Hayes et al. 1996, Taniguchi and Nakano 2000, Dunham et al. 2003) is well established. At a smaller scale, the distribution of individual fish species within a watershed are expected to follow patterns similar to those over the species geographic range (Taniguchi et al. 1998, Taniguchi and Nakano 2000).

Occupancy rates for sites within the core range of most species were high in the upper basin (Table 7). Most species occupied all sites that fell within their core range, suggesting that core-range accurately captured the range of basin area over which the species is likely to be persistent in the absence of disturbance. Species with low core occupancy rates were relatively rare and the ability to develop an accurate core range may have been hampered by a lack of presences or individuals. Occupancy rates in periphery sites were generally lower. However, ubiquitous core species had high

periphery occupancy rates supporting the idea that the species are relatively evenly distributed over the entire upper basin.

Presence/absence models behaved similarly to the core area models and provided further evidence to support the core area models I developed. Classification strength of the DFA presence/absence models for species in the upper basin generally categorized both predicted presences and absences correctly greater than 80% of the time.

Individual fish species were effective in understanding the importance of local and regional factors and the importance of watershed connectivity. Mottled sculpin were present in every upper basin site and were the most numerically dominant species in most streams (Freund unpublished data). Low relative proportion of mottled sculpin in the Cheat River was an effective indicator of local impairment. Brook trout proved to be an excellent indicator of local and regional conditions in the upper Cheat River. Brook trout are classified as an intolerant species (McCormick et al. 2001) and were among the most widely distributed species in the upper basin. Their distribution ranged from the smallest to the largest sites sampled in the upper basin. Brook trout in the Appalachians have been shown to spawn in streams much smaller than the smallest site sampled in this study (3.2 km²; Lamothe 2002). The occurrence of brook trout in intermediate and large streams may be an indicator of watershed connectivity. Lastly, examination of periphery occupancy rates of intermediate and large stream core species provided another indicator of regional conditions. Relatively high occupancy rates within the periphery of these species' core ranges may indicate strong watershed connectivity and the importance of dispersal in periphery occupancy rates.

Local and Regional Impairment in the Lower Cheat River Basin

Degraded water quality in the lower basin overwhelmed models developed in the upper basin and applied to the lower basin. The Cheat River watershed has a history of AMD impairment (WVDEP 1996, Williams 1997) and sites without fishes are responsible for much of the deviation from upper basin patterns. However, with fishless sites removed, the relationship is altered due the effects of degraded water chemistry and the resultant loss of watershed connectivity. Severely and moderately impaired sites were important in assessing the local impacts of AMD. All severely impaired and several moderately impaired sites had no fishes present. Moderately impaired sites displayed considerable species richness deviation when compared to the upper basin. Intolerant species as well as many tolerant species were absent from moderately impaired sites, very likely due to the toxic effects of heavy metals (Atland and Barlaup 1996, Baldigo and Lawrence 2000, Barry et al. 2000, Beaumont et al. 2000). In addition to low species richness, moderately impaired sites also tended to have high proportions of tolerant species, a common response to aquatic stressors (Karr 1981, 19991, McCormick et al. 2001).

Local impairment due to mining-related discharges in the lower Cheat River basin is prevalent (Freund Chapter 1, WVDEP 1996). A greater proportion of moderate to large streams are impaired and these intermediate and large streams comprise the core habitat for most species in the Cheat River (Figure 8). Reduction or elimination biological integrity of these intermediate and large streams systems has far-reaching impacts as discussed later.

Species that were predicted to be present but were not present were often intolerant or non-tolerant species that were expected to be absent due to local water quality impairment (Welsh and Perry 1997, McCormick 2001). In the lower basin, blacknose dace and creek chubs were the two most commonly present species and both are generally considered to be tolerant to pollution (McCormick et al. 2001). Similarly, white sucker, another tolerant species, was the only species that occurred in more of the lower basin sites that contained fish than it did in the upper basin. Intolerant species such as brook trout were often absent from lower basin stream reaches in which they were predicted to be present.

Species richness patterns and occupancy rates in non-AMD impacted lower basin sites are expected to be indicators of regional conditions. Observed species richness was predicted with 56% accuracy in unimpaired lower basin sites compared to 79% using the same model for upper basin sites. Only two unimpaired lower basin sites had observed/expected values greater than expected value of one (Figure 6). Similarly, core and periphery occupancy (Table 9) and occupancy of sites that the presence/absence model predicted the correctly (Table 11) were generally much lower for unimpaired lower basin sites than for upper basin sites. Together these results provide evidence of a regional impairment in the lower basin.

In the lower Cheat River basin the greatest positive deviation in species richness was generally associated with the degraded condition of nearby streams or was associated with Cheat Lake, a large impoundment of the Cheat River. Moderately AMD impacted stream reaches near the Cheat River, particularly Roaring Creek which confluent with a thermally-impacted and moderately AMD-impacted section of the Cheat River, likely

provided temporal refuge habitat (Schlosser 1995a) despite slight water chemistry impairment. This pattern was likely exacerbated during the low-flow conditions under which fish were sampled as AMD is pervasive in groundwater that makes up much of the summer baseflow (Petty and Barker 2004) and thermal effluent is most likely to exceed many fishes incipient maximum thermal tolerance (Kilgour et al. 1985). A similar pattern was observed in Daugherty Run, a moderately impaired stream approximately 2 km upstream of Roaring Creek but above the thermal discharge and low head dam. The Cheat River is moderately impaired by AMD at both sites. Although positive deviation in observed against expected species richness is generally considered to be indicative of connectivity (Oberdorff et al. 2001), regional impairment in the lower basin may be responsible for increased richness in these sample sites.

Fishes inhabiting an intermediate stream core appear to be most impacted by the regional effects of AMD but generalizations are difficult within this group. In the upper basin, these species are widely distributed and often among the most abundant of species in many sites. Within the lower basin, these species are generally rare and are associated with streams that are connected to the mainstem of the Cheat River or Big Sandy Creek. Northern hogsuckers, river chubs, and central stonerollers, the three most widely distributed intermediate stream core species in the upper basin, are extremely rare in the lower basin (Table 2). Occupancy rates for both core and periphery sites were greatly reduced in the lower basin. Additionally, occupancy rates of lower basin sites where species were predicted to be present by presence/absence models were also greatly reduced. These same general patterns held true for both moderately and unimpaired lower basin sites. Within the Muddy Creek subwatershed, these species are absent and

likely extirpated due to AMD impacts within a large portion of their core range. These impacts limit the amount of habitat core area available, reducing the probability of persistence (Donalson and Nisbet 1999) and the probability that they are found in periphery sites. Similarly, in the Little Sandy Creek drainage, these species are rare or extirpated through much of what is considered to be their expected range.

Mechanisms of Regional Impairment

The effects of mining-related discharge are prevalent in lower Cheat River basin (Freund Chapter 1, WVDEP 1996, Williams et al. 1999, Petty and Barker 2004). At a local scale, the impacts of AMD on biotic communities are easily assessed and detected (Freund Chapter 1, Herlihy et al. 1990, WVDEP 1996). Less easily detected or understood is how multiple local impairments are manifested to produce a measurable regional impairment.

I propose that regional impairments are produced through the disruption of watershed connectivity that limits dispersal, isolates stream reaches, removes or reduces access to complementary and supplementary habitats (Dunning et al. 1992, Schlosser 1995a), and reduces the potential of colonizing species. Mining drainage may create potential barriers that reduce or eliminate dispersal altering ecological processes in non-impaired stream reaches and isolate unimpaired stream reaches. Secondly, AMD disrupts demographic process in stream reaches that are both directly impaired reaches and adjacent non-impaired reaches by reducing supplementary or complementary habitats (Schlosser 1995a). This disruption of demographic processes is likely to eliminate or reduce dispersal of individuals from these directly and indirectly impaired reaches.

Through these mechanisms, AMD, which is generally viewed as a local stressor, reduces the overall biotic integrity of the lower Cheat River basin.

The lower Cheat River basin is fragmented by two distinct forms of chemical barriers. The lower reaches of several tributaries are severely chemically impaired and are likely to be absolute barriers to fish dispersal. For example, from May 2002 through May 2003, the pH sampled at two locations in the lower 5 km of Muddy Creek never exceeded 5.0 and rarely exceeded 4.0. Combined with high toxic metal concentrations, fishes are likely to avoid this acutely toxic dispersal corridor (Atland and Barlaup 1996, Woodward et al. 1997). The effects of isolation by severely impacted stream reaches are analogous to disruption in connectivity associated with dams (Winston et al. 1991, Bednarek 2001) and produced similar results in this study.

A second type of barrier is temporal and analogous to Schlosser's (1995a) semi-permeable barrier. A temporal barrier to dispersal exists when moderately impaired streams, which vary temporally in acidity and toxic metal concentrations (Petty and Barker 2004), act as barriers to dispersal. Travel costs are likely to be highly different among different species (Bernstein et al. 1991), depending upon dispersal ability and their tolerance to different water chemistry conditions.

The effects of barriers on species' distribution are likely to vary predictably based on individual species life history characteristics and their core range and extent that is impacted. Species tolerant to chemical stressors, particularly those that are highly mobile species, are more likely to persist within a degraded stream reach and disperse from or through the degraded stream reach (Schlosser 1995a). Intolerant or non-tolerant species were shown to be absent or greatly reduced in moderately impaired sites and

consequently moderately impaired sites are reduced in their ability to serve as a source of colonizing individuals (Osborne and Wiley 1992). Stream reaches in the lower Cheat River basin above AMD sources show a loss of species richness, evident by several notable missing species in isolated or recovering stream reaches, examples of which were discussed earlier. Isolation may be partially responsible for species absences as isolation has been shown to reduce species persistence (e.g. Winston et al. 1991, Hilderbrand and Kershner 2000, Harig and Fausch 2002).

Although the mine impairment in the lower Cheat River basin provided a unique model system, these same mechanisms have been observed in other impacted watersheds. Similar regional impairments may be expected in impounded (Winston et al. 1991, Fausch and Young 1995, Porto et al. 1999), urbanized (Wang et al. 2000, 2003), deforested (Nislow and Lowe 2003), and acid precipitation impacted (Driscoll et al. 2001) watersheds.

Assessing Regional Impairment

Although no single measure may capture watershed scale impairment, a strong case for impairment can be produced by combining information regarding the deviation from species richness, core-periphery, and individual species presence/absence models developed in reference watersheds.

Despite some inherent limitations, core range proved to be a useful tool that incorporated life history characteristics into a simple measure of the central tendency of individual species distribution. An examination of individual fish species core range provides a useful starting point to begin to estimate the impacts of anthropogenic disturbance of fish distribution and relative abundance. The core-periphery view

establishes a framework by which both local and regional impairments may be examined. Results from the core-periphery models were backed by the species richness and presence/absence models providing convincing evidence of a regional impairment.

Core range is likely a conservative estimate of the range of basin area which are important to individual species, is not necessarily analogous to source-sink dynamics (Pulliam 1988), and does not necessarily encompass reproductive or important supplementary habitats. As it is a measure of the center of each species' accumulation of individuals as a function of basin area, estimates of core range are necessarily biased towards the center of the distribution of basin areas sampled. For example, brook trout in the Appalachians spawn in streams with basin areas much smaller than the smallest stream I sampled (3.16 km²; Lamothe 2002). The core range of brook trout is likely overestimated and not necessarily analogous to source habitats for brook trout. Conversely, large river fishes' core range was likely an underestimate as much of their core larger than my range of wadeable streams. The development of core ranges were based on sites sampled during the summers of 2002 and 2003. However, fish distribution and relative abundance are spatially and temporally variable (Horwitz 1978, Schlosser 1985, Grossman et al. 1998, Schlosser and Kallenmeyn 2000). Sampling occurred during summer low flow periods during a time that did not coincide with spawning activity for many fishes and is generally regarded as a period when fish communities are the most stable (Barbour et al. 1999).

An understanding of life history characteristics of stream fishes, particularly basin area associations, water chemistry and sediment tolerance, and dispersal ability provides a useful framework to assess local and regional impacts on stream fish assemblages and

individual fish species. Unfortunately our understanding of the basic life history and ecology of many stream fishes, even those that are relatively common, hampers our ability to understand their response to anthropogenic stressors, particularly the indirect impacts these disturbances may produce. Examination of the patterns of individual species distribution, even those which are not generally considered indicator species, provided an important means of assessing regional impairment.

Patterns displayed by missing species and in species recovery generally follow existing models. Reduced connectivity and barriers, whether complete or semi-permeable (Schlosser 1995a), create isolated populations where recolonization is greatly reduced or eliminated. If fish populations exist as metapopulations, reduction of connectivity and reduction in patch size caused by AMD is more likely to result in localized extirpation of individual species (Pulliam 1988, Schlosser 1998). This is evident with many species in the lower Cheat Basin and in particular several recovering subwatersheds within the lower basin are missing species historically present and expected to be present based on patterns in the non-impacted upper basin.

Presence/absence and density of individual fish species provided a great deal of insight about how life history of individual species and impacts to their core habitat areas have had larger, regional effects on fish assemblages. Missing species have many common characteristics. Most species missing from lower basin sites are intolerant to pollution or at least non-tolerant (McCormick et al. 2001), many are limited in dispersal ability in the case of mottled sculpin (Petty and Grossman 2004) or are highly mobile and regional impairment may be linked to their extirpation such as the case of brook trout

(Riley et al. 1992, Gowan et al. 1994) and smallmouth bass (Todd and Rabeni 1989), or the sampled reach is outside of their core range.

Ubiquitously distributed species follow two distinct patterns. They are often regarded as pioneering species and are classified as tolerant species in many biomonitoring programs (blacknose dace and creek chub); McCormick et al. 2001, Pirhalla 2004). However, our view of species tolerance is confounded by a lack of life history information particularly concerning dispersal and susceptibility across a range of potential physical and chemical stressors. These species, particularly when they are present in high proportions, may be indicative of not only local but regional impairment. A second pattern in ubiquitously distributed species is displayed by mottled sculpin and longnose dace which are relatively immobile benthic fishes (Mullen and Burton 1995, Petty and Grossman 2004) that may have complementary habitats distributed within a relatively small area. Thus, these species may be less affected by isolation but display responses to local water quality.

Unlike commonly used assessment tools like fish based index of biotic integrity (IBI; Karr 1981, 1991, McCormick et al. 2001), assessment of regional impairment does not provide a single quantifiable deviation from reference conditions. Rather assessment of regional impairment relies upon interpreting information regarding the deviation from expected species richness and individual species distribution patterns. This approach requires an understanding of life history of fishes and interpretation of results that may provide conflicting information. The assessment of regional impairment was straightforward and obvious in the highly impaired lower Cheat River basin. However, the assessment relies heavily upon an understanding of life history characteristics and

professional judgment. Additionally, since watershed impairment is not captured by a single measure and is not yet well defined, assessing when a watershed becomes impaired requires judgment and weighing of the information provided coupled with knowledge of ecological processes and species life history characteristics.

Fish Species as Indicators of Regional Impairment

Fishes are commonly used as indicators of water quality (e.g. Pinder and Morgan 1995, Welsh and Perry 1997, Baldigo and Lawrence 2000) or habitat degradation (e.g. Gorman and Karr 1978, Pinder and Morgan 1995, Pirhalla 2004). This association between fish species and their physical and chemical environment is the basis of the Index of Biotic Integrity (Karr 1981, 1991). However, these efforts have been largely focused on local scales. Little effort has been exerted to determine species that are indicators of regional conditions. I present three complementary views of species with characteristics that make them viable candidates to assess regional impairment.

First, relatively immobile but widely distributed species provide a measure of watershed-scale impairment. Their limited dispersal ability will make them slow or unable to repopulate extirpated stream reaches. In the Cheat River watershed, mottled sculpin (*Cottus bairdi*) provide a potentially excellent indicator of regional impairment. Mottled sculpin provide an excellent indicator of local and historic water chemistry conditions as they are non-tolerant (McCormick et al. 2001), are ubiquitous distributed over the range of wadeable streams, and are relatively immobile (Petty and Grossman 2004). Whereas they were the most prevalent and most numerous species in the unimpacted upper basin, deviation from this pattern in the lower basin is indicative of local and possible regional impairment. Mottled sculpin are a relatively long-lived

benthic invertivore that is generally relatively immobile with most individuals occupying small home ranges (Petty and Grossman 2004). Absence of sculpin or very low densities is indicative of local impairment (McCormick et al. 2001) or regional and historic impairment. Due to their relatively low dispersal range (Petty and Grossman 2004), sculpin are likely slow to recolonize sites where they were historically extirpated, providing evidence of historical degradation.

Second, relatively mobile species that are not tolerant to pollution provide an indication of watershed connectivity and thus potential watershed-scale impairment. In contrast to mottled sculpin, smallmouth bass are top predators in many lotic ecosystems and are capable of frequent long dispersal events (Todd and Rabeni 1989) in search of seasonal habitats or forage. Smallmouth bass generally inhabit larger streams and rivers (Schneberger 1972). In the upper basin, smallmouth bass presence was strongly positively correlated with basin area (71.6%), although they were generally found in low densities in small streams near the Cheat River which acted as supplementary habitats (Schlosser 1995a). The presence of smallmouth bass, determined through a discriminant function model, was correlated with basin area in the upper basin. However, in the lower basin smallmouth bass were present in only three of the 14 sites (21.4%) of the sites where they were expected to be present. The only three lower basin sites in which they were collected all were near the confluence with large rivers that likely served as the source population (Pulliam 1988). Smallmouth bass occupy a large home range (Todd and Rabeni 1989) and loss of connectivity to supplementary habitats and a reduction in suitable area within their home range is likely to have lead to local extirpation in the lower Cheat River basin.

Smallmouth bass provide information about regional and historic impairment evident in the lower Cheat River watershed. Little Sandy Creek is recovering from AMD impairment and falls well within the range of stream sizes that predict smallmouth bass presence from the upper basin. Smallmouth bass were historically extirpated from the drainage and now that the stream is recovering, a physical barrier prevents fishes within Big Sandy Creek from recolonizing the Little Sandy Creek drainage and other similarly impacted and recovering stream in the lower Cheat River basin.

Similar to smallmouth bass, brook trout are a relatively mobile (Riley et al. 1992, Gowan et al. 1994) small stream species that may also provide information about watershed connectivity. Brook trout were most numerous in small, coldwater streams. They were the only small stream core species in the Cheat River and provided a unique opportunity to test the effects of watershed connectivity. They are generally competitively dominant in small streams (Taniguchi et al. 1998), and general spawn in small headwater stream (Brasch et al. 1973, Lamothe 2002). Much recent research investigating brook trout dispersal indicates that they are much more mobile than previously thought (Riley et al. 1992, Gowan et al. 1994). Their presence in larger streams and rivers is tied to tributary density or other localized inputs of coldwater (Torgersen et al. 1999) and through dispersal from small stream source populations (Pulliam 1988).

Brook trout provide some of the best evidence for regional impairment in the lower Cheat River basin. Brook trout were among the most commonly occurring species in the lower basin and were present in low densities within larger streams in the upper Cheat River basin, including the largest site sampled (141.02 km²). However, in the

lower basin, brook trout were only found in 33% of lower basin sites that were not fishless. Many of the lower basin absences were linked to impaired water chemistry but they are also missing from unimpaired sites. Streams of larger basin area may provide important supplementary habitats (Schlosser 1995a) that provide improved foraging potential (West Virginia University, Unpublished data) and degradation in these moderate to large basin area streams may be partially responsible for upstream extirpation. Regional impairment in the lower basin is evident through the loss of brook trout in the Little Sandy Creek watershed. Beaver Creek, a large tributary to Little Sandy Creek is treated for AMD near the headwaters. Brook trout were unable to recolonize the headwaters of Beaver Creek due to a lack of nearby source populations and were transplanted by the West Virginia Division of Natural Resources (WVDNR) and a local Trout Unlimited group.

Lastly, fishes that are often categorized as “pioneering” species (Smith 1971) such as creek chub and green sunfish can be indicators of regional impairment. In addition to these species, other tolerant species such as blacknose dace and white suckers are potential indicators of local and regional impairment. Tolerant species are expected to be present in many sites and several were categorized as ubiquitously distributed in the upper Cheat River basin. However, when tolerant species constitute a large proportion of total number of individuals present within a reach, they are used as an indicator of impairment (e.g. McCormick et al. 2001). However, I propose that aside from their tolerance to degraded conditions, tolerant species may become relatively common in streams within an impaired watershed. First, many of these species are highly mobile as the term “pioneering” suggests (Smith 1971). Secondly, these species may be more

tolerant to local degradation and thus may serve as the only source of colonizing species from moderately impaired stream reaches.

This pattern was evident in many subwatershed in the lower Cheat River basin. Muddy Creek provides an excellent example of the potential for tolerant and pioneering species to be indicators of regional impairment. The lower reaches of Muddy Creek are among the most AMD-impaired within the Cheat River basin and likely the United States. Upstream, a section of moderately impaired stream exists which is devoid of many species predicted to be present based on patterns evident in the upper basin. Northern hogsuckers, central stonerollers, and river chubs are among the conspicuously absent species in this section Muddy Creek. Upstream of Muddy Creek's first source of AMD the stream is among the least impacted in the Cheat River basin. Additionally, limestone geology in the upper Muddy Creek drainage sustains excellent water quality and macroinvertebrate populations (WVDEP 1996). However, I propose that as a result of the downstream extirpation of northern hogsuckers, central stonerollers, and river chubs and a disruption to these species' core area, they are absent from upstream sites with excellent water quality. Consequently, tolerant species may be present in high proportions in non-impaired stream reaches within impaired watersheds as a result of a lack of competition and the lack of colonizing individuals of other species. This pattern is also evident in several other subwatersheds within the Cheat River basin.

Implications of Regional Impairment

Currently, degradation of aquatic resources is viewed as a local phenomenon and is assessed at a local, stream reach scale (e.g. WV SCI, Tetra Tech 2000; MAHA IBI, McCormick et al. 2001). Generally results from a single sample are viewed to be

representative of ecological conditions at the reach or segment scale. Watershed impairment is difficult to assess using current methods and at best may be estimated through average multimetric index scores compared to reference conditions (e.g. McCormick et al. 2001). However, this approach is unlikely to differentiate local and regional sources of impairment. Viewing impairment at the watershed level fits well with the prevailing interest in watershed and ecosystem management (Reichman and Pulliam 1996, Roper et al. 1997) and has the potential to have implications for bioassessment and restoration programs.

To better meet the objectives of the Clean Water Act (1972) that our nation's waterways support "the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water", a broader view of how local impacts affect large-scale ecological processes should be incorporated into the assessment of aquatic resources. Local impacts that disrupt the rates of demographic processes, reduce connectivity and species dispersal, and eliminate intolerant and non-tolerant species may affect stream reaches which are not directly impacted, producing a regional impact. This view of regional impairment is not currently recognized under state and federal regulatory programs or recognized in the prioritization of AMD treatment in watershed restoration programs. Additionally, regional reference conditions are determined through examining what are considered least-impacted reference reaches. However, species richness and composition in these reaches may be unknowingly compromised through regional impairment. Incorporation of regional impairments in watershed assessment programs would provide a more robust and meaningful representation of the impacts of stressors on aquatic ecosystems.

Many effective indices of biological condition exist (Karr 1981, TetraTech 2000, McCormick et al. 2001) and have been shown to effectively capture the effects anthropogenic degradation in the Cheat River basin (Freund Chapter 1). However, fish assemblages provide a measure of regional biological condition which can not be ascertained using water chemistry or macroinvertebrate assemblages (Freund Chapter 1). I propose that the core-periphery view and examination of regional impacts established in this study provides a useful, common sense approach to assessing the biological conditions at the watershed scale.

Implications for Watershed Restoration

Restoration of the lower Cheat River basin requires an understanding of two important interacting mechanisms that will limit the impacts of localized AMD treatment. First, AMD isolates stream reaches by reducing or eliminating dispersal between stream reaches. Secondly, current and historic water chemistry has reduced the number of species within many subwatershed which reduces the “near-local” source of colonizing individuals. Disruption of the species-area relationship and loss of individual species within the lower Cheat River basin demonstrate how local AMD impacts create a regional impact. As demonstrated, recovery in the lower Cheat River has been slowed by barriers to dispersal or slowed by lingering water chemistry impairment.

Prioritization of AMD treatment in the lower Cheat River basin has been based on improving water chemistry. Currently, streams targeted for restoration are among the most inexpensive and simplest to recover to water quality standards. Current regulations and assessment of treatment focuses on obtaining water chemistry improvements. Pollutant regulation and the newly imposed TMDL on the Cheat River focus on reach

scale impairments and do not consider region-wide impacts of pollutants. Additionally, TMDL monitoring uses water chemistry measures while the CWA mandated watershed assessments use macroinvertebrates to measure human impacts (WVDEP 1996). Neither of these measures are sensitive to regional impairments (Freund Chapter 1) and produce a short-term view of localized impacts.

Ongoing recovery in much of the lower Cheat River basin is evident as are limitations in colonization due to chemical barriers, historic loss of species, and dispersal ability. Recovery is dependent upon dispersal ability (Phinney 1975, Ensign et al. 1997), presence of physical or chemical barriers (Ensign et al. 1997, Warren and Pardew 1998), impacts relative to core habitat area, and current water chemistry conditions. Disruption or fragmentation of areas within a species' core range may reduce its potential recovery. Despite improvements in water chemistry, recovery may be immaterial if a nearby source of colonizing individuals is not present which the case in the lower Cheat River basin is often. Recolonization of isolated stream reaches may be enhanced through species reintroductions (Lonzarich et al. 1998) and supplementation (Hilderbrand 2002). These options may be necessary in many recovering streams in the Cheat River basin as several waterfalls form natural barriers in the lower Cheat basin.

Treatment of AMD has greatly improved the ecological condition of individual streams in the lower Cheat River basin; however regional impairment is still evident. Current restoration has been focused on stream reaches which are easily treated with little consideration given to effects the treatment may have on surrounding stream reaches. When possible, projects should recognize that restoration of connectivity can have regional effects on stream fish communities. Restoration is expensive and often cost

prohibitive (WVDEP 1996). These limited resources need to be prioritized so that treatment may improve regional biological conditions.

Conclusions

Within the lower Cheat River basin, localized impairments have a regional effect on fish assemblages and individual fish species. While there is no single measure of watershed scale impairment, alteration of the species-area relationship, predictable presence/absence of individual fish species, and the generally reduced ecological condition (Freund Chapter 1, Williams et al. 1999) in the lower Cheat River basin are indicative of multiple local impairments that are manifested at larger spatial scales, creating a subwatershed that is impaired as a whole. Although AMD provided a unique model system, many other sources of aquatic degradation are likely to operate in a similar manner. As an example, reduced habitat complexity, channelization, stormsewer runoff, and other potential sources of degradation associated with urbanization (Wang et al. 2000, 2003) are likely to have similar regional effects.

Although many researchers and resource managers embrace watershed and ecosystem management units (Reichman and Pulliam 1996, Roper et al. 1997), assessment and monitoring programs often focus on local processes. Little consideration has been given to how stream reaches interact with one another or how regional dynamics influence processes at the stream reach, segment, or drainage basin scales (Fausch et al. 2002). In order to better assess watershed scale impairment, an improved knowledge and understanding of fish life history characteristics and how they interact with their environment over spatial and temporal scales is needed.

References

- Aadland, L. P. 1993. Stream habitat types: their fish assemblages and relationship to flow. *North American Journal of Fisheries Management* 13:790-806.
- Angermeier, P. L. and I. J. Schlosser. 1989. Species-area relationship for stream fishes. *Ecology* 70:1450-1462.
- Angermeier, P. L. and M. R. Winston. 1998. Local vs. regional influences on local diversity in stream fish communities of Virginia. *Ecology* 79:911-927.
- Atland, A. and B. T. Barlaup. 1996. Avoidance behaviour of Atlantic Salmon (*Salmo salar* L.) fry in waters of low pH and elevated aluminum concentrations: laboratory experiments. *Canadian Journal of Fisheries and Aquatic Sciences* 53:1827-1834.
- Baldigo, B. P. and G. B. Lawrence. 2000. Composition of fish communities in relation to stream acidification and habitat in the Neversink River, New York. *Transactions of the American Fisheries Society* 129:60-76.
- Baltz, D. M. and P. B. Moyle. 1993. Invasion resistance to introduced species by a native assemblage of California stream fishes. *Ecological Applications* 3:246-255.
- Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.
- Barry, K. L., J. A. Grout, C. D. Levings, B. H. Nidle, and G. E. Piercey. 2000. Impacts of acid mine drainage on juvenile salmonids in an estuary near Britannia Beach in Howe Sound, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences* 57:2032-2043.
- Beaumont, M. W., P. J. Butler, and E. W. Taylor. 2000. Exposure of brown trout, *Salmo trutta*, to a sub-lethal concentration of copper in soft acidic water: effects upon muscle metabolism and membrane potential. *Aquatic Toxicology* 51(2):259-272.
- Bednarek, A.T. 2001. Undamming rivers: a review of the ecological impacts of dam removal. *Environmental Management* 27:803-814.
- Bernstein, C., A. Kacelnik, and J. R. Krebs. 1991. Individual decisions and the distribution of predators in a patchy environment. II. The influence of travel costs and structure of the environment. *Journal of Animal Ecology* 60:205-225.
- Bilby, R. E. 1984. Characteristics and frequency of cool-water areas in western Washington streams. *Journal of Freshwater Ecology* 2:593-602.

- Brasch, J., J. McFadden, and S. Kmiotek. 1973. Brook Trout: Life history, ecology, and management. Wisconsin Department of Natural Resources Publication 226.
- Brown, J. H., G. C. Stevens, and D. M. Kaufman. 1996. The geographic range: Size, shape, boundaries, and internal structure. *Annuals Review of Ecology and Systematics* 27:597-623.
- Cardwell, D. H., R. B. Erwin, H. P. Woodward, and C. W. Lotz. 1968. Geologic map of West Virginia. West Virginia Geologic and Economic Survey.
- Clesceri, L. S., Greenberg, A.E., Eaton, A.D., eds. 1992. Standard Methods for the Examination of Water and Wastewater, 18th Edition. American Public Health Association, Washington, D.C.
- Cornell, H. V. and J. H. Lawton. 1992. Species interactions, local and regional processes, and limits to the richness of ecological communities: a theoretical perspective. *Journal of Animal Ecology* 61:1-12.
- Curry, K. D. and A. Spacie. 1984. Differential use of stream habitat by spawning catostomids. *American Midland Naturalist* 111:267-279.
- Dewalle, D. R., W. E. Sharpe, and P. J. Edwards. 1988. Biogeochemistry of two Appalachian deciduous forest sites in relation to episodic stream acidification. *Water, Air, and Soil Pollution* 40:143-156.
- Donalson, D. D. and R. M. Nisbet. 1999. Population dynamics and spatial scale: effects of system size on population persistence. *Ecology* 80:2492-2507.
- Driscoll, C. T., G. B. Lawrence, A. J. Bulger, T. J. Butler, C. S. Cronan, C. Eagar, K. F. Lambert, G. E. Likens, J. L. Stoddard, and K. C. Weathers. 2001. Acidic deposition in the Northeastern United States: Source and inputs, ecosystem effects, and management strategies. *BioScience* 51:180-198.
- Dunham, J. B., G. L. Vinyand, and B. E. Rieman. 1997. Habitat fragmentation and extinction risk of Lahontan cutthroat trout. *North American Journal of Fisheries Management* 17:1126-1133.
- Dunham, J., B. Rieman, and G. Chandler. 2003. Influences of temperature and environmental variables on the distribution of bull trout within streams at the southern margin of its range. *North American Journal of Fisheries Management* 23:894-904.
- Dunning, J. B., B. J. Danielson, and H. R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. *Oikos* 65:169-175.
- Ensign, W. E., K. N. Leftwich, P. L. Angermeier, and C. A. Dolloff. 1997. Factors influencing stream fish recovery following a large-scale disturbance. *Transactions of the American Fisheries Society* 126:895-907.

- Fausch, K. D. and M. K. Young. 1995. Evolutionarily significant units and movement of resident stream fishes: a cautionary tale. *American Fisheries Society Symposium* 17:360-370.
- Fausch, K. D., C. E. Torgersen, C. V. Baxter, and H. W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience* 52:483-498.
- Federal Water Pollution Control Act. 1972. Public Law 107-303.
- Felley, J. D. and L. G. Hill. 1983. Multivariate assessment of environmental preferences of cyprinid fishes of the Illinois River, Oklahoma. *American Midland Naturalist* 109:209-221.
- Flebbe, P. A. 1994. A regional view of the margin: salmonid abundance and distribution in the Southern Appalachian Mountains of North Carolina and Virginia. *Transactions of the American Fisheries Society* 123:657-667.
- Galbreath, P. F., N. D. Adams, S. Z. Guffey, C. J. Moore, and J. L. West. 2001. Persistence of native Southern Appalachian brook trout populations in the Pigeon River system, North Carolina. *North American Journal of Fisheries Management* 21:927-934.
- Gaston, K. J., T. M. Blackburn, J. J. D. Greenwood, R. D. Gregory, R. M. Quinn, and J. H. Lawton. 2000. Abundance-occupancy relationships. *Journal of Applied Ecology* 37:39-59.
- Gerking, S. D. 1953. Evidence for the concepts of home range and territory in stream fishes. *Ecology* 34:347-365.
- Gore, J. A. and A. M. Milner. 1990. Island biogeographical theory: can it be used to predict lotic recovery rates? *Environmental Management* 14:737-753.
- Gorman, O. T. and J. R. Karr. 1978. Habitat structure and stream fish communities. *Ecology* 59:507-515.
- Gorman, O. T. 1986. Assemblage organization of stream fishes: the effect of rivers on adventitious streams. *The American Naturalist* 128:611-616.
- Gowan, C., M. K. Young, K. D. Fausch, and S. C. Riley. 1994. Restricted movement in resident stream salmonids: a paradigm lost. *Canadian Journal of Fisheries and Aquatic Sciences* 51:2626-2637.
- Grenouillet, G., D. Pont, and C. Hérissé. 2004. Within-basin fish assemblage structure: the relative influence of habitat versus stream spatial position on local species richness. *Canadian Journal of Fisheries and Aquatic Sciences* 61:93-102.

- Grossman, G. D., Jr. R. E. Ratajczak, M. Crawford, and M. C. Freeman. 1998. Assemblage organization in stream fishes: effects of environmental variation and interspecific interactions. *Ecological Monographs* 68:395-420.
- Guegan, J. F., S. Let, and T. Oberdorff. 1998. Energy availability and habitat heterogeneity predict global riverine fish diversity. *Nature* 391:382-384.
- Harig, A. L., and K. D. Fausch. 2002. Minimum habitat requirements for establishing translocated cutthroat trout populations. *Ecological Applications* 12:535-551.
- Hayes, D. B., C. P. Ferreri, and W. W. Taylor. 1996. Linking fish habitat to their population dynamics. *Canadian Journal of Fisheries and Aquatic Sciences* 53:383-390.
- Hilderbrand, R. H. and J. L. Kershner. 2000. Conserving inland cutthroat trout in small streams: how much stream is enough? *North American Journal of Fisheries Management* 20:13-520.
- Hilderbrand, R. H. 2002. Simulation supplementation strategies for restoring and maintaining stream resident cutthroat trout populations. *North American Journal of Fisheries Management* 22(3):879-887.
- Herlihy, A. T., P. R. Kaufmann, M. E. Mitch, and D. D. Brown. 1990. Regional estimates of acid mine drainage impact on streams in the Mid-Atlantic and Southeastern United States. *Water, Air, and Soil Pollution* 50:91-107.
- Horwitz, R. J. 1978. Temporal variability patterns and the distributional patterns of stream fishes. *Ecological Monographs* 48:307-321.
- House, R. 1995. Temporal variation in abundance of an isolated population of cutthroat trout in Western Oregon, 1981-1991. *North American Journal of Fisheries Management* 15:33-41.
- Jackson, D. A. and H. H. Harvey. 1989. Biogeographic associations in fish assemblages: local vs. regional processes. *Ecology* 70:1472-1484.
- Kanehl, P. D., J. Lyons, and J. E. Nelson. 1997. Changes in the habitat and fish community of the Milwaukee River, Wisconsin, following removal of the Woolen Mills dam. *North American Journal of Fisheries Management* 17:387-400.
- Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6:91-107.
- Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management. *Ecological Applications* 1:66-84.

- Kilgour, D. M., R. W. McCauley, and W. Kwain. 1985. Modeling the lethal effects of high temperature on fish. *Canadian Journal of Fisheries and Aquatic Sciences* 42(5):947-951.
- Kinsolving, A. D. and M. B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. *Ecological Applications* 3:531-544.
- Kwak, T. J. and T. F. Waters. 1997. Trout production dynamics and water quality in Minnesota streams. *Transactions of the American Fisheries Society* 126:35-48.
- Labbe, T. R. and K. D. Fausch. 2000. Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. *Ecological Applications* 10:1774-1791.
- Lamothe, P. J. 2002. Spatial population dynamics of brook trout (*Salvelinus fontinalis*) in a central Appalachia watershed. Master's Thesis, West Virginia University, Morgantown, WV.
- Lee, D. C. and R. E. Rieman. 1997. Population viability assessment of salmonids by using probabilistic networks. *North American Journal of Fisheries Management* 17:1144-1157.
- Levin, S. A. 1992. The problem of pattern and scale in Ecology. *Ecology* 73:1943-1967.
- Lonzarich, D. G., Jr. M. L. Warren, and M. R. E. Lonzarich. 1998. Effects of habitat isolation on the recovery of fish assemblages in experimentally defaunated stream pools in Arkansas. *Canadian Journal of Fisheries and Aquatic Sciences* 55:2141-2149.
- Lotrich, V.A. 1973. Growth, production, and community composition of fishes inhabiting a first-, second-, and third-order stream of Eastern Kentucky. *Ecological Monographs* 43:377-397.
- Lydersen, E., N. W. A. Rukke, J. G. B. Jensen, B. M. Kjelsberg, B. Tørnsjø, R. D. Vogt, L. A. Vøllestad, and A. B. S. Poleo. 2002. Seasonal variation in mortality of brown trout (*Salmo trutta*) in an acidic aluminum-rich lake. *Journal of Limnology* 61:61-68.
- Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. *North American Journal of Fisheries Management* 12:198-203.
- MacArthur, R. and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, NJ.
- Matthews, W. J. and H. W. Robison. 1998. Influence of drainage connectivity, drainage area, and regional species richness on fishes of the interior highlands in Arkansas. *The American Midland Naturalist* 139:1-19.

- McCormick, F. H., R. M. Hughes, P. R. Kaufmann, D. V. Peck, J. L. Stoddard, and A. T. Herlihy. 2001. Development of an index of biotic integrity for the Mid-Atlantic Highland Region. *Transactions of the American Fisheries Society* 130:857-877.
- McGarigal, K., S. Cushman, and S. Stafford. 2000. *Multivariate Statistics for Wildlife and Ecology Research*. Springer-Verlag, New York, 283p.
- Meador, M. R., J. P. McIntyre, and K. H. Pollock. 2003. Assessing the efficacy of single-pass backpack electrofishing to characterize fish community structure. *Transactions of the American Fisheries Society* 132(1):39-46.
- Meffe, G. K. 1984. Effects of abiotic disturbance on coexistence of predator-prey fish species. *Ecology* 65:1525-1534.
- Mendelson, J. 1975. Feeding relationships among species of *Notropis* (Pisces: Cyprinidae) in a Wisconsin stream. *Ecological Monographs* 45:199-232.
- Minshall, G.W., R. C. Petersen, K. W. Cummins, T. L. Bott, J. R. Sedell, C. E. Cushing, and R. L. Vannote. 1983. Interbiome comparison of stream ecosystem dynamics. *Ecological Monographs* 53:1-25.
- Mullen, D. M. and T. M. Burton. 1995. Size-related habitat use by longnose dace (*Rhinichthys cataractae*). *American Midland Naturalist* 133: 177-183.
- Munn, N. L. and J. L. Meyer. 1990. Habitat-specific solute retention in two small streams: an intersite comparison. *Ecology* 71:2069-2082.
- Newall, P. R., and J. J. Magnuson. 1999. The importance of ecoregion versus drainage area on fish distributions in the St. Croix River and its Wisconsin tributaries. *Environmental Biology of Fishes* 55(3):245-54.
- Nislow, K. H. and W. H. Lowe. 2003. Influences of logging history and stream pH on brook trout abundance in first-order streams in New Hampshire. *Transactions of the American Fisheries Society* 132:166-171.
- Oberdorff, T., B. Hugueny, and J-F. Guegan. 1997. Is there an influence of historical events on contemporary fish species richness in rivers? Comparisons between Western Europe and North America. *Journal of Biogeography* 24:461-467.
- Oberdorff, T., D. Pont, B. Hugueny, and D. Chessel. 2001. A probabilistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. *Freshwater Biology* 46:399-415.
- Ohio Environmental Protection Agency. 1989. *Biological criteria for the protection of aquatic life. Vol. III. Users manual for biological field assessment of Ohio surface waters*. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, OH.

- Osborne, L. L. and M. J. Wiley. 1988. Empirical relationships between land use/cover and stream water quality in an agricultural watershed. *Journal of Environmental Management* 26:9-27.
- Osborne, L. L. and M. J. Wiley. 1992. Influence of tributary spatial position on the structure of warmwater fish communities. *Canadian Journal of Fisheries and Aquatic Sciences* 49:671-681.
- Parsons, M., M. C. Thomas, and R. H. Norris. 2003. Scales of macroinvertebrate distribution in relation to the hierarchical organization of river systems. *Journal of the North American Benthological Society* 22:105-122.
- Peterson, J. T., and C. F. Rabeni. 2001. The relation of fish assemblages to channel units in an Ozark stream. *Transactions of the American Fisheries Society* 130:911-926.
- Petty, J. T., and J. Barker. 2004. Water quality variability, trace metals and implications for restoring a mined Appalachian watershed. *Proceedings of the National Meeting of the American Society of Mining and Reclamation*. ASMR, Lexington, KY.
- Petty, J. T. and G. D. Grossman. 2004. Restricted movement by mottled sculpin (pisces: cottidae) in a southern Appalachian stream. *Freshwater Biology* 49:631-645.
- Phinney, D. E. 1975. Repopulation of an eradicated stream section by brook trout. *Transactions of the American Fisheries Society* 104:685-687.
- Pinder, M. J. and I. R. P. Morgan. 1995. Interactions of pH and habitat on Cyprinid distributions in Appalachian streams of Maryland. *Transactions of the American Fisheries Society* 124:94-102.
- Pirhalla, D. E. 2004. Evaluating fish-habitat relationships for refining regional indexes of biotic integrity: development of a tolerance index of habitat degradation for Maryland stream fishes. *Transactions of the American Fisheries Society* 133:144-159.
- Poff, N. L. and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. *Ecology* 76:606-327.
- Porto, L. M., R. L. McLaughlin, and D. L. G. Noakes. 1999. Low-head barrier dams restrict the movements of fishes in two Lake Ontario streams. *North American Journal of Fisheries Management* 19:1028-1036.
- Post, D. M., J. F. Kitchell, and J. R. Hodgson. 1998. Interactions among adult demography, spawning date, growth rate, predation, overwinter mortality, and the recruitment of largemouth bass in a northern lake. *Canadian Journal of Fisheries and Aquatic Sciences* 55:2588-2600.

- Pulliam, H.R. 1988. Sources, sinks, and population regulation. *American Naturalist* 132:652-661.
- Rahel, F. J. 2000. Homogenization of fish faunas across the United States. *Science* 288:854-856.
- Rathert, D., D. White, J. C. Sifneos, and R. M. Hughes. 1999. Environmental correlates of species richness for native freshwater fish in Oregon, U.S.A. *Journal of Biogeography* 26:257-273.
- Reichmann, O. J. and H. R. Pulliam. 1996. The scientific basis for ecosystem management. *Ecological Applications* 6:694-696.
- Riley, S. C., K. D. Fausch, and C. Gowan. 1992. Movements of brook trout (*Salvelinus fontinalis*) in four small subalpine streams in northern Colorado. *Ecology of Freshwater Fish* 1:112-122.
- Roper, B. B., J. J. Dose, and J. E. Williams. 1997. Stream restoration: is fisheries biology enough? *Fisheries* 22:6-11.
- Schaeffer, J. F. and J. R. Kerfoot. 2004. Fish assemblage dynamics in an adventitious stream: a landscape perspective. *American Midland Naturalist* 151:134-145.
- Schlosser, I. J. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. *Ecology* 66:1484-1490.
- Schlosser, I. J. 1987a. A conceptual framework for fish communities in small warmwater streams. *In*: Matthews, W.J. and D. C. Heins (editors), *Community and evolutionary ecology of North American stream fishes*, University of Oklahoma Press, Norman, OK.
- Schlosser, I. J. 1987b. The role of predation in age- and size-related habitat use by stream fishes. *Ecology* 68:651-659.
- Schlosser, I. J. and K. K. Ebel. 1989. Effects of flow regime and cyprinid predation on a headwater stream. *Ecological Monographs* 59:41-57.
- Schlosser, I. J. 1991. Stream fish ecology: A landscape perspective. *BioScience* 41:704-712.
- Schlosser, I. J. 1995a. Critical landscapes attributes that influence fish population dynamics in headwater streams. *Hydrobiologia* 303:71-81.
- Schlosser, I. J. 1995b. Dispersal, boundary processes, and trophic-level interactions in streams adjacent to beaver ponds. *Ecology* 76:908-925.

- Schlosser, I. J. and P. L. Angermeier. 1995. Spatial variation in demographic processes of lotic fishes: conceptual models, empirical evidence, and implications for research. *American Fisheries Society Symposium* 17:392-401.
- Schlosser, I. J. 1998. Fish recruitment, dispersal, and trophic interactions in a heterogeneous lotic environment. *Oecologia* 113:260-268.
- Schlosser, I. J. and L. W. Kallenmeyn. 2000. Spatial variation in fish assemblages across a beaver-influenced successional landscape. *Ecology* 81:1371-1382.
- Schmetterling, D. A. 2003. Reconnecting a fragmented river: Movements of Westslope cutthroat trout and bull trout after transport upstream of Milltown Dam, Montana. *North American Journal of Fisheries Management* 23:721-731.
- Schneberger, E. 1972. Smallmouth Bass: Life History, ecology, and management. Wisconsin Department of Natural Resources Publication 242.
- Scheurer, J. A., K. D. Fausch, and K. R. Bestgen. 2003. Multiscale processes regulate brassy minnow persistence in a Great Plains river. *Transactions of the American Fisheries Society* 132:840-855.
- Scullion, J., and R. W. Edwards. 1980. The effect of pollutants from the coal industry on the fish fauna of a small river in the South Wales coalfield. *Environmental Pollution* 21(2):141-153.
- Sheldon, A. L. 1968. Species diversity and longitudinal succession in stream fishes. *Ecology* 49:193-198.
- Smith, P. W. 1971. Illinois streams: a classification based on their fishes and an analysis of factors responsible for the disappearance of native species. Illinois Natural History Survey Biological Notes, 76.
- Smithson, E. B. and C. E. Johnston. 1999. Movement patterns of stream fishes in a Ouachita Highlands stream: an examination of the restricted movement paradigm. *Transactions of the American Fisheries Society* 128:847-853.
- Stacey, P. B. and M. Taper. 1992. Environmental variation and the persistence of small populations. *Ecological Applications* 2:18-29.
- Stauffer, J. R.Jr., J. M. Boltz, and L. R. White. 1995. *The Fishes of West Virginia*, Academy of Natural Sciences, Philadelphia.
- Strager, M. P. 2002. Watershed Characterization and Modeling System Version 2.8. Natural Resources Analysis Center, West Virginia University. WVDEP Water Resources Contract # 0704-0188.

- Strange, R. M. and B. M. Burr. 1997. Intraspecific phylogeography of North American highland fishes: a test of the Pleistocene vicariance hypothesis. *Evolution* 51: 885-897.
- Taniguchi, Y., F. J. Rahel, D. C. Novinger, and K. G. Gerow. 1998. Temperature mediation of competitive interactions among three fish species that replace each other along longitudinal stream gradients. *Canadian Journal of Fisheries and Aquatic Sciences* 55:1894-1901.
- Taniguchi, Y. and S. Nakano. 2000. Condition-specific competition: implications for the altitudinal distribution of stream fishes. *Ecology* 81:2027-2039.
- Taylor, C. M. and M. L. Warren Jr. 2001. Dynamics in species composition of stream fish assemblages: environmental variability and nested subsets. *Ecology* 82:2320-2330.
- Tetra Tech, Inc. 2000. A Stream Condition Index for West Virginia Wadeable Streams. Owings Mills, MD.
- Thurrow, R. F., D. C. Lee, and B. E. Rieman. 1997. Distribution and status of seven native salmonids in the interior Columbia River basin and portions of the Klamath River and Great Basins. *North American Journal of Fisheries Management* 17:1094-1110.
- Todd, B. L. and C. F. Rabeni. 1989. Movement and habitat use by stream-dwelling Smallmouth bass. *Transactions of the American Fisheries Society* 118:229-242.
- Tonn, W. J., J. J. Magnuson, M. Rask, and J. Toivonen. 1990. Intercontinental comparison of small-lake fish assemblages: the balance between local and regional processes. *The American Naturalist* 136:345-375.
- Torgersen, C. E., D. M. Price, H. W. Li, and B. A. McIntosh. 1999. Multiscale thermal refugia and stream associations of chinook salmon in Northeastern Oregon. *Ecological Applications* 9:301-319.
- Vadas, R. L. and D. J. Orth. 2000. Habitat use of fish communities in a Virginia stream system. *Environmental Biology of Fishes* 59:253-569.
- Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37:130-137.
- Wang, L., J. Lyons, P. Kanehl, R. Bannerman, and E. Emmons. 2000. Watershed urbanization and changes in fish communities in Southeastern Wisconsin streams. *Journal of the American Water Resources Association* 36: 1173-1189.

- Wang, L., J. Lyons, and P. Kanehl. 2003. Impacts of urban land cover on trout streams in Wisconsin and Minnesota. *Transactions of the American Fisheries Society* 132: 825-839.
- Ward, J. V. 1989. Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation* 83:269-278.
- Warren, M. L. Jr. and M. G. Pardew. 1998. Road crossings as barriers to small-stream fish movement. *Transactions of the American Fisheries Society* 127:637-644.
- Waters, T. F., J. P. Kaehler, T. J. Polomis, and T. J. Kwak. 1993. Production dynamics of smallmouth bass in a small Minnesota stream. *Transactions of the American Fisheries Society* 122:588-598.
- Wear, D. N., M. G. Turner, and R. J. Naiman. 1998. Land cover along an urban-rural gradient: Implications for water quality. *Ecological Applications* 8:619-630.
- Wehrly, K. E., M. J. Wiley, and P. W. Seelbach. 2003. Classifying regional variation in thermal regime based on stream fish community patterns. *Transactions of the American Fisheries Society* 132:18-38.
- Welsh, S. A. and S. A. Perry. 1997. Acidification and fish occurrence in the upper Cheat River drainage, West Virginia. *Journal of the American Water Resources Association* 33:423-429.
- Williams, D. R., M. E. Clark, and J. B. Brown. 1999. Stream water quality in coal mined areas of the lower Cheat River basin, West Virginia and Pennsylvania, during low-flow conditions, July 1997. *Water Resource Investigations Report* 98-4258 8.
- Winston, M. R., C. M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. *Transactions of the American Fisheries Society* 120:98-105.
- Woodward, D. F., J. N. Goldstein, A. M. Farag, and W. G. Brumbaugh. 1997. Cutthroat trout avoidance of metals and conditions characteristic of a mining waste site: Coeur d' Alene River, Idaho. *Transaction of the American Fisheries Society* 126: 699-706.
- WVDEP. 1996. An ecological assessment of the Cheat River watershed. West Virginia Division of Environmental Protection. Report 0502004 – 1996.
- Yoder, C. O. and M. A. Smith. 1999. Using fish assemblages in a state biological assessment and criteria program: essential concepts and considerations. Pages 17-56 in T. P. Smith, editor. *Assessing the sustainability and biological integrity of water resources using fish communities*. CRC Press, Boca Raton, Florida.

Zar, J. H. 1999. Biostatistical Analysis: Fourth Edition. Prentice Hall, Upper Saddle River, New Jersey.

Table 1. Principal component analysis factor loadings of Cheat River watershed water chemistry data.

	PC1	PC2
eigenvalue	7.35	2.87
Proportion of Variance Explained	0.52	0.20
pH	-0.78	0.52
Sp. Conductance	0.78	0.51
TDS	0.75	0.35
Alkalinity	-0.46	0.78
Acidity	0.50	-0.65
Sulfate	0.89	0.33
Al	0.91	-0.30
Cd	--	--
Cr	0.33	--
Fe	0.88	--
Mn	0.91	--
Ni	0.94	--
Calcium Hardness	0.55	0.76
Total Hardness	0.75	0.55

Table 2. Occurrence of the most common species in the upper and lower basins of the Cheat River. For each the upper, lower, and entire Cheat River basin, the number of sampled stream reaches the species occurred in, the percentage of reaches it occurred in, and the relative rank for the number of sites the species occurred within is given. There were 27 upper basin sites, 37 lower basin sites, and 27 lower basin sites with fishes present. Species are: blacknose dace (RHCA), mottled sculpin (COBA), creek chub (SEAT), longnose dace (RHCA), brook trout (SAFO), white sucker (CACO), greenside darter (ETBL), central stoneroller (CAAN), northern hogsucker (HYNI), river chub (NOMI), green sunfish (LECY), smallmouth bass (MIDO), and rainbow trout (ONMY). Species are listed in order of percent occurrence in the Cheat River watershed as a whole.

	Upper Basin			Lower Basin				Cheat Basin		
	# Sites Captured	% Occurrence	Rank Occurrence	# Sites Captured	% Occurrence	Rank Occurrence	% Occurrence in Sites with Fish	Cheat Basin	Cheat Basin % Occurrence	Cheat Basin Rank
RHAT	25	93%	2	23	62%	1	85%	48	75%	1
COBA	27	100%	1	19	51%	3	70%	46	72%	2
SEAT	22	81%	4	22	59%	2	81%	44	69%	3
RHCA	25	93%	2	9	24%	7	33%	34	53%	4
SAFO	22	81%	4	9	24%	7	33%	31	48%	5
CACO	13	48%	11	15	41%	4	56%	28	44%	6
ETBL	15	56%	9	10	27%	5	37%	25	39%	7
CAAN	18	67%	6	7	19%	11	26%	25	39%	7
HYNI	15	56%	9	8	22%	9	30%	23	36%	9
NOMI	17	63%	7	5	14%	12	19%	22	34%	10
LECY	11	41%	12	10	27%	5	37%	21	33%	11
MIDO	16	59%	8	3	8%	13	11%	19	30%	12
ONMY	5	19%	13	8	22%	9	30%	13	20%	13

Table 3. Mean proportion of tolerant and intolerant species (McCormick et al. 2001) for each site in the upper and lower basins as well as lower basin sites that contained fish.

	Upper Basin	Lower Basin	Lower Basin in Sites with Fish
Mean Proportion Tolerant	0.20	0.41	0.58
Standard Error	0.03	0.07	0.07
Mean Proportion Intolerant	0.07	0.07	0.10
Standard Error	0.02	0.03	0.04

Table 4. Discriminant function analysis classification/misclassification analysis using model built based on upper basin presence/absence data applied to lower basin to predict species occurrence. Percent values represent percentage of each present or absent species that were correctly or incorrectly classified within the lower basin. Only species for which a significant model could be constructed are presented.

Species Code		Upper Basin				Lower Basin			
		Absent and Predicted to be:		Present and Predicted to be:		Absent and Predicted to be:		Present and Predicted to be:	
		Absent	Present	Absent	Present	Absent	Present	Absent	Present
AMRU	N	9	1	2	7	20	3	3	3
	%	90%	10%	22%	78%	87%	13%	50%	50%
CAAN	N	3	2	4	10	9	15	1	4
	%	60%	40%	29%	71%	38%	63%	20%	40%
CACO	N	10	2	1	6	6	12	0	11
	%	83%	17%	14%	86%	33%	67%	0%	100%
CLFU	N	12	1	1	5	22	6	1	0
	%	92%	8%	17%	83%	79%	21%	100%	0%
ETBL	N	8	0	1	10	15	9	1	4
	%	100%	0%	9%	100%	63%	38%	20%	80%
ETFL	N	8	3	2	6	19	7	0	3
	%	73%	27%	25%	75%	73%	27%	0%	100%
HYBRID	N	13	1	1	4	25	4	0	0
	%	93%	7%	20%	80%	86%	14%	0%	0%
HYNI	N	7	1	0	11	9	14	3	3
	%	88%	13%	0%	100%	39%	61%	50%	50%

LECY	N	12	1	1	5	19	2	6	2
	%	92%	8%	17%	83%	91%	10%	75%	25%
MAMA	N	12	3	0	4	27	2	0	0
	%	80%	20%	0%	100%	93%	7%	0%	0%
MIDO	N	8	0	1	10	15	11	0	3
	%	100%	0%	9%	91%	58%	42%	0%	100%
NOMI	N	5	1	2	11	7	18	2	2
	%	83%	17%	15%	85%	28%	72%	50%	50%
Notropis spp.	N	13	0	0	6	21	4	3	1
	%	100%	0%	0%	100%	84%	16%	75%	25%
SAFO	N	3	0	0	16	8	12	4	5
	%	100%	100%	100%	100%	40%	60%	44%	56%
SEAT	N	3	0	1	15	2	10	4	13
	%	100%	0%	6%	94%	17%	83%	24%	77%
Total	N	123	14	13	116	215	114	27	50
	%	90%	10%	10%	90%	65%	35%	35%	65%

Table 5. Variables included in stepwise discriminant models developed from upper basin site data.

Species	Variable 1	Partial R ²	Variable 2	Partial R ²	Variable 3	Partial R ²	Variable 4	Partial R ²	Total Model Ave. Squared Canonical Correlation
AMRU	Shreve Order	0.59	--	--	--	--	--	--	0.59
CAAN	Basin Area	0.23	--	--	--	--	--	--	0.23
CACO	--	--	--	--	--	--	--	--	--
CLFU	Shreve Order	0.28	--	--	--	--	--	--	0.28
COBA	--	--	--	--	--	--	--	--	--
ETBL	Basin Area	0.56	RVHA Score	0.23	Strahler Order	0.13	--	--	0.70
ETFL	Basin Area	0.22	Strahler Order	0.19	--	--	--	--	0.38
Hybrid	Basin Area	0.32	Distance from Mainstem	0.24	--	--	--	--	0.48
HYNI	Basin Area	0.43	Strahler Order	0.16	--	--	--	--	0.53
LECY	Basin Area	0.52	--	--	--	--	--	--	0.52
MAMA	Shreve Order	0.22	--	--	--	--	--	--	0.22
MIDO	Basin Area	0.57	--	--	--	--	--	--	0.57
NOMI	Basin Area	0.30	--	--	--	--	--	--	0.30
Notropis	Basin Area	0.66	RVHA Score	0.16	--	--	--	--	0.71
RHAT	--	--	--	--	--	--	--	--	--
RHCA	--	--	--	--	--	--	--	--	--
SAFO	Strahler Order	0.17	RVHA Score	0.15	D-Link	0.16	Distance from Mainstem	0.39	0.64
SEAT	Basin Area	0.18	Shreve Order	0.14	--	--	--	--	0.30

Table 6. Number of correctly predicted presences, predicted presences, and percent of correct predictions from upper basin derived presence/absence models for all lower basin sites, unimpaired and moderately impaired lower basin sites, and upper basin sites.

	Correctly Predicted Presences	Total Predicted Occupancies	Percent Correctly Predicted Occupancy	Correctly Predicted Presences	Total Predicted Occupancies	Percent Correctly Predicted Occupancy
	Lower Basin All Sites			Lower Basin Unimpaired Sites		
AMRU	3	6	50.0%	0	1	0.0%
CAAN	4	19	21.1%	2	7	28.6%
CACO	11	25	44.0%	5	10	50.0%
CLFU	0	6	0.0%	0	1	0.0%
COBA	15	29	51.7%	10	12	83.3%
ETBL	4	13	30.8%	1	3	33.3%
ETFL	3	10	30.0%	2	4	50.0%
HYN1	3	17	17.6%	1	7	14.3%
LECY	2	4	50.0%	0	0	N/A
MAMA	0	2	0.0%	0	1	0.0%
MIDO	3	14	21.4%	2	4	50.0%
NOMI	2	20	10.0%	1	8	12.5%
NOTROPIS	1	5	20.0%	0	1	0.0%
RHAT	17	29	58.6%	8	12	66.7%
RHCA	7	29	24.1%	5	12	41.7%
SAFO	5	17	29.4%	3	7	42.9%
SEAT	13	23	56.5%	7	10	70.0%
	Lower Basin Moderately Impaired			Upper Basin		
AMRU	3	4	75.0%	7	9	77.8%
CAAN	2	8	25.0%	10	14	71.4%
CACO	6	9	66.7%	6	7	85.7%
CLFU	0	5	0.0%	5	6	83.3%
COBA	5	13	38.5%	27	27	100.0%
ETBL	3	7	42.9%	10	11	90.9%
ETFL	1	4	25.0%	6	8	75.0%
HYN1	2	8	25.0%	11	11	100.0%
LECY	2	3	66.7%	5	6	83.3%
MAMA	0	1	0.0%	4	4	100.0%
MIDO	1	7	14.3%	10	11	90.9%
NOMI	1	9	11.1%	11	13	84.6%
NOTROPIS	1	3	33.3%	6	6	100.0%
RHAT	9	13	69.2%	25	27	92.6%
RHCA	2	13	15.4%	25	27	92.6%
SAFO	2	7	28.6%	16	16	100.0%
SEAT	6	10	60.0%	15	15	100.0%

Table 7. Variables included in stepwise discriminant models developed from lower basin site data.

Species	Variable 1	Partial R ²	Variable 2	Partial R ²	Variable 3	Partial R ²	Variable 4	Partial R ²	Total Model Ave. Squared Canonical Correlation
AMRU	Basin Area	0.17	--	--	--	--	--	--	0.17
			Distance from						
CAAN	Basin Area	0.26	Mainstem	0.18	WQ Factor 1	0.14	--	--	0.47
CACO	Basin Area	0.26	WQ Factor 1	0.16	--	--	--	--	0.38
CLFU	D-Link	0.10	--	--	--	--	--	--	0.10
COBA	Strahler Order	0.27	WQ Factor 1	0.35	RVHA Score	0.10	--	--	0.58
ETBL	Shreve Order	0.40	--	--	--	--	--	--	0.39
ETFL	D-Link	0.32	0.0934	--	--	--	--	--	0.38
					Link Order				
HYNI	Basin Area	0.36	WQ Factor 1	0.09	Difference	0.16	--	--	0.51
LECY	Basin Area	0.28	WQ Factor 2	0.14	Shreve Order	0.09	--	--	0.44
	Distance from								
MIDO	Mainstem	0.25	--	--	--	--	--	--	0.25
NOMI	D-Link	0.24	Basin Area	0.85	WQ Factor 1	0.09	--	--	0.37
Notropis	D-Link	0.24	Basin Area	0.09	--	--	--	--	0.37
RHAT	WQ Factor 2	0.46	WQ Factor 1	0.17	Strahler Order	0.11	--	--	0.61
RHCA	Strahler Order	0.15	WQ Factor 1	0.11	--	--	--	--	0.25
SAFO	WQ Factor 1	0.13	RVHA Score	0.10	Shreve Order	0.10	Strahler Order	0.17	0.41
SEAT	WQ Factor 2	0.27	Strahler Order	0.30	WQ Factor 1	0.22	--	--	0.60

Table 8. Discriminant function analysis classification/missclassification results for lower basin sites using models developed from the lower basin.

Species Code		Lower Basin			
		Absent and Predicted to be:		Present and Predicted to be:	
		Absent	Present	Absent	Present
AMRU	N	19	4	0	6
	%	83%	17%	0%	100%
CAAN	N	22	2	0	5
	%	92%	8%	0%	100%
CACO	N	16	2	1	10
	%	89%	11%	9%	91%
CLFU	N	22	6	0	1
	%	79%	21%	0%	100%
COBA	N	10	4	0	15
	%	71%	29%	0%	100%
ETBL	N	23	1	0	5
	%	96%	4%	0%	100%
ETFL	N	23	3	0	3
	%	88%	12%	0%	100%
HYNI	N	21	2	0	6
	%	91%	9%	0%	100%
LECY	N	17	4	1	7
	%	81%	19%	13%	88%
MIDO	N	25	1	0	3
	%	96%	4%	0%	100%
NOMI	N	22	3	0	4
	%	88%	12%	0%	100%
Notropis spp.	N	22	3	0	4
	%	88%	12%	0%	100%
RHAT	N	11	1	0	17
	%	92%	8%	0%	100%
RHCA	N	17	5	1	6
	%	77%	23%	14%	86%

SAFO	N	18	2	1	8
	%	90%	10%	11%	89%
SEAT	N	9	3	0	17
	%	75%	25%	0%	100%
Total	N	297	46	4	117
	%	87%	13%	3%	97%

Table 9. Variables included in stepwise discriminant models developed from unimpaired lower basin sites.

Species	Variable 1	Partial R ²	Variable 2	Partial R ²	Variable 3	Partial R ²	Variable 4	Partial R ²	Variable 5	Partial R ²	Total Model Ave. Squared Canonical Correlation
AMRU	--	--	--	--	--	--	--	--	--	--	--
CAAN	--	--	--	--	--	--	--	--	--	--	--
CACO	--	--	--	--	--	--	--	--	--	--	--
CLFU	Distance from Mainstem	0.56	--	--	--	--	--	--	--	--	0.56
COBA	WQ Factor 2	0.62	Strahler Order	0.49	Basin Area	0.48	WQ Factor 1	0.40	Distance from Mainstem	0.51	0.97
ETBL	Distance from Mainstem	0.56	--	--	--	--	--	--	--	--	0.56
ETFL	D-Link	0.40	--	--	--	--	--	--	--	--	0.40
HYNI	Distance from Mainstem	0.56	--	--	--	--	--	--	--	--	0.56
LECY	Distance from Mainstem	0.56	--	--	--	--	--	--	--	--	0.56
MIDO	Basin Area	0.56	Shreve Order	0.29	--	--	--	--	--	--	0.69
NOMI	D-Link	0.40	--	--	--	--	--	--	--	--	0.40
Notropis	D-Link	0.40	--	--	--	--	--	--	--	--	0.40
RHAT	WQ Factor 1	0.49	RVHA Score	0.48	D-Link	0.22	--	--	--	--	0.83
RHCA	WQ Factor 1	0.29	Strahler Order	0.30	WQ Factor 2	0.28	--	--	--	--	0.64
SAFO	--	--	--	--	--	--	--	--	--	--	--
SEAT	WQ Factor 2	0.31	--	--	--	--	--	--	--	--	0.31

Table 10. Variables included in stepwise discriminant models developed from moderately impaired lower basin sites.

Species	Variable 1	Partial R ²	Variable 2	Partial R ²	Variable 3	Partial R ²	Variable 4	Partial R ²	Total Model Ave. Squared Canonical Correlation
AMRU	Basin Area	0.52	Distance from Mainstem	0.35	--	--	--	--	0.69
CAAN	Basin Area	0.26	Distance from Mainstem	0.18	WQ Factor 1	0.14	--	--	0.47
CACO	Strahler Order	0.71	Basin Area	0.25	--	--	--	--	0.78
CLFU	Distance from Mainstem	0.28	WQ Factor 1 Link Order	0.23	Shreve Order Distance from Mainstem	0.35	D-Link	0.25	0.73
COBA	Shreve Order	0.60	Difference	0.60	--	--	--	--	0.82
ETBL	Shreve Order	0.60	--	--	--	--	--	--	0.60
ETFL	Distance from Mainstem	0.60	RVHA Score Distance from Mainstem	0.48	D-Link	0.22	--	--	0.83
HYNI	Basin Area	0.52	--	--	--	--	--	--	0.69
LECY	Basin Area	0.41	--	--	--	--	--	--	0.41
MIDO	Distance from Mainstem	0.60	RVHA Score	0.48	D-Link	0.22	--	--	0.83
NOMI	D-Link	0.41	Basin Area	0.33	Strahler Order	0.23	--	--	0.83
Notropis	D-Link	0.41	Basin Area	0.33	Strahler Order	0.23	--	--	0.69
RHAT	WQ Factor 2	0.53	Basin Area	0.38	Shreve Order	0.29	--	--	0.79
RHCA	Distance from RVHA Score	0.34	Mainstem	0.47	--	--	--	--	0.65
SAFO	Strahler Order	0.21	--	--	--	--	--	--	0.21
SEAT	Strahler Order	0.61	WQ Factor 2	0.38	--	--	--	--	0.76

Table 11: Discriminant function analysis classification/missclassification results for lower basin models developed for unimpaired and moderately impaired sites within the lower basin.

Species Code		Unimpaired				Moderately Impaired			
		Absent and Predicted to be:		Present and Predicted to be:		Absent and Predicted to be:		Present and Predicted to be:	
		Absent	Present	Absent	Present	Absent	Present	Absent	Present
AMRU	N	9	0	0	2	10	0	0	4
	%	100%	0%	0%	100%	100%	0%	0%	100%
CAAN	N	11	0	0	3	9	0	0	2
	%	100%	0%	0%	100%	100%	0%	0%	100%
CACO	N	7	0	0	7	6	1	0	4
	%	100%	0%	0%	100%	86%	14%	0%	100%
CLFU	N	13	0	0	1	--	--	--	--
	%	100%	0%	0%	100%	--	--	--	--
COBA	N	2	0	0	9	8	0	0	6
	%	100%	0%	0%	100%	100%	0%	0%	100%
ETBL	N	9	0	0	2	11	0	0	3
	%	100%	0%	0%	100%	100%	0%	0%	100%
ETFL	N	10	0	0	1	12	0	0	2
	%	100%	0%	0%	100%	100%	0%	0%	100%
HYNI	N	9	0	0	2	10	0	0	4
	%	100%	0%	0%	100%	111%	0%	0%	100%

LECY	N	9	0	0	2	6	2	0	6
	%	100%	0%	0%	100%	75%	25%	0%	100%
MIDO	N	10	0	0	2	12	0	0	2
	%	100%	0%	0%	100%	100%	0%	0%	100%
NOMI	N	10	0	0	1	11	0	0	3
	%	100%	0%	0%	100%	100%	0%	0%	100%
Notropis spp.	N	11	0	0	3	10	0	0	1
	%	100%	0%	0%	100%	100%	0%	0%	100%
RHAT	N	4	0	0	7	4	0	0	10
	%	100%	0%	0%	100%	100%	0%	0%	100%
RHCA	N	7	0	0	4	11	0	0	3
	%	100%	0%	0%	100%	100%	0%	0%	100%
SAFO	N	4	0	0	7	12	0	0	2
	%	100%	0%	0%	100%	100%	0%	0%	100%
SEAT	N	4	0	0	7	4	0	--	--
	%	100%	0%	0%	100%	100%	0%	--	--
Total	N	129	0	0	60	136	3	0	52
	%	100%	0%	0%	100%	98%	2%	0%	100%

Table 12. Basin area (km²) at the 25th, 50th, and 75th percentile of relative cumulative proportion of individuals for commonly occurring fish species in the upper Cheat River basin, their core extent of their core range (difference between 25th and 75th percentiles). Max D was determined using a Kolmogorov-Smirnov goodness of fit for continuous data test. Values exceeding the expected value of 0.25908 have distributions that differ from expectations.

Species	25th Percentile	50th Percentile	75th Percentile	Core Range	Max D	Core Type	Core Extent
SAFO	6.3	9.3	15.1	8.8	0.619	Small	Narrow
CACO	20.4	31.6	45.7	25.3	0.429	Intermediate	Narrow
RHAT	14.8	38.0	70.8	56.0	0.192	Ubiquitous	Wide
CLFU	35.5	41.7	47.9	12.4	0.453	Intermediate	Narrow
SEAT	33.9	45.7	51.3	17.4	0.252	Ubiquitous	Narrow
MAMA	15.1	47.9	64.6	49.4	0.301	Intermediate	Narrow
AMRU	32.4	52.5	95.5	63.1	0.333	Intermediate	Wide
LECY	44.7	55.0	66.1	21.4	0.346	Intermediate	Narrow
RHCA	29.5	55.0	81.3	51.8	0.162	Ubiquitous	Wide
COBA	26.3	56.2	91.2	64.9	0.190	Ubiquitous	Wide
NOMI	45.7	61.7	87.1	41.4	0.335	Intermediate	Narrow
HYBRID	45.7	64.6	81.3	35.6	0.351	Intermediate	Narrow
MIDO	51.3	72.4	91.2	39.9	0.369	Large	Narrow
HYNI	33.1	74.1	87.1	54.0	0.338	Intermediate	Wide
ETBL	30.2	79.4	91.2	61.0	0.329	Large	Wide
CAAN	56.2	87.1	117.5	61.3	0.401	Large	Wide
ETFL	53.7	87.1	125.9	72.2	0.497	Large	Wide
NOTROPIS	85.1	102.3	123.0	37.9	0.690	Large	Narrow
EXPECTED							
LINE	20.0	45.7	81.3	61.3	0.25908	--	

Table 13. Percentage of sites in the lower and upper basins within the core and periphery area ranges where each commonly occurring species was present. Upper – lower basin represents the difference in the percentage of core and periphery sites that were occupied by each species. Large differences suggest species that were commonly absent in the lower basin whereas smaller differences show distribution more similar to the upper basin. Negative values represent species that were more commonly present in the lower basin than in the upper basin.

	Upper Basin		Lower Basin		Upper - Lower Basin	
	% Core Sites Occupied	% Periphery Sites Occupied	% Core Sites Occupied	% Periphery Sites Occupied	% Core Sites Occupied Difference	% Periphery Sites Occupied Difference
AMRU	87.5	31.6	20.0	17.9	67.5	13.7
CAAN	100.0	57.1	40.0	15.2	60.0	42.0
CACO	60.0	40.9	77.8	31.0	-17.8	9.9
CLFU	100.0	36.0	33.3	0.0	66.7	36.0
COBA	100.0	100.0	63.6	44.4	36.4	55.6
ETBL	100.0	33.3	36.4	11.1	63.6	22.2
ETFL	66.7	28.6	0.0	12.5	66.7	16.1
HYBRID	40.0	31.8	0.0	3.1	40.0	28.7
HYNI	100.0	40.0	40.0	14.3	60.0	25.7
LECY	75.0	34.8	60.0	21.2	15.0	13.6
MAMA	30.0	11.8	0.0	5.0	30.0	6.8
MIDO	100.0	50.0	20.0	6.1	80.0	43.9
NOMI	100.0	52.4	14.3	12.9	85.7	39.5
NOTROPIS	100.0	32.0	50.0	11.1	50.0	20.9
RHAT	100.0	87.5	73.7	47.4	26.3	40.1
RHCA	100.0	85.0	50.0	14.3	50.0	70.7
SAFO	85.7	88.9	23.1	28.0	62.6	60.9
SEAT	100.0	79.2	80.0	57.6	20.0	21.6
AVERAGE	85.8	51.2	37.9	19.6	47.9	31.5

Table 14. Number and percent of core and periphery sites occupied for commonly occurring species within non-AMD impacted sites in the lower Cheat River basin compared to upper basin core and periphery occupancy. Upper basin core and periphery occupancy rates are provided for comparison.

	Upper Basin		Core Sites Occupied	Core Sites Sampled	Lower Basin			
	% Core Sites Occupied	% Periphery Sites Occupied			% Core Sites Occupied	Periphery Sites Occupied	Periphery Sites Sampled	% Periphery Sites Occupied
AMRU	87.5%	31.6%	0	2	0.0%	2	12	16.7%
CAAN	100.0%	57.1%	1	1	100.0%	2	13	15.4%
CACO	60.0%	40.9%	1	3	33.3%	4	11	36.4%
CLFU	100.0%	36.0%	0	0	N/A	0	14	0.0%
COBA	100.0%	100.0%	2	2	100.0%	8	12	66.7%
ETBL	100.0%	33.3%	2	2	100.0%	1	12	8.3%
ETFL	66.7%	28.6%	0	1	0.0%	2	13	15.4%
HYBRID	40.0%	31.8%	0	2	0.0%	0	12	0.0%
HYN1	100.0%	40.0%	2	2	100.0%	1	12	8.3%
LECY	75.0%	34.8%	2	2	100.0%	1	12	8.3%
MAMA	30.0%	11.8%	0	0	N/A	0	14	0.0%
MIDO	100.0%	50.0%	1	2	50.0%	0	12	0.0%
NOMI	100.0%	52.4%	1	2	50.0%	1	12	8.3%
NOTROPIS	100.0%	32.0%	0	0	N/A	1	14	7.1%
RHAT	100.0%	87.5%	6	7	85.7%	3	7	42.9%
RHCA	100.0%	85.0%	2	2	100.0%	3	12	25.0%
SAFO	85.7%	88.9%	5	7	71.4%	3	7	42.9%
SEAT	100.0%	79.2%	0	0	N/A	9	14	64.3%
Total	85.8%	51.2%	25	37	67.6%	41	215	19.1%

Table 15. Summary of core types, ecological attributes and representative species in the Cheat River watershed.

Core Type	Description	Cheat Basin Examples
Ubiquitous Core	<ul style="list-style-type: none"> • Presence and relative abundance is not associated with basin area • Spawning activity widely distributed • May be persistent in isolated stream reaches 	<ul style="list-style-type: none"> • <i>Semotilus atromaculatus</i> • <i>Cottus bairdi</i> • <i>Rhinichthys atratulus</i> • <i>Rhinichthys cataractae</i>
Small Stream Core	<ul style="list-style-type: none"> • Spawning activity limited or dominant in small streams • Relative abundance much greater in small streams • Presence in larger streams linked to coldwater inputs; supplemental habitat • Persistent in small, isolated stream reaches 	<ul style="list-style-type: none"> • <i>Salvelinus fontinalis</i>
Intermediate Stream Core	<ul style="list-style-type: none"> • Spawning activity in intermediate basin area streams • Rarely sampled in headwater streams • May not persist upstream of isolation 	<ul style="list-style-type: none"> • <i>Nocomis micropogon</i> • <i>Hypentelium nigricans</i> • <i>Campostoma anomalum</i>
Large River Core	<ul style="list-style-type: none"> • Large river species • Distribution limited to larger basin areas and well connected streams • May use intermediate and small streams as supplementary habitats 	<ul style="list-style-type: none"> • <i>Micropterus dolomieu</i> • Large river minnows (<i>Notropis</i> sp., <i>Cyprinella</i> sp., <i>Luxilus</i> sp.) • Large river sucker species (<i>Moxostoma</i> sp.)

Figure Legend

Figure 1. Fish sample sites in the upper and lower basin of the Cheat River, West Virginia.

Figure 2. Water chemistry PCA factor scores for the Cheat River Basin.

Figure 3. Stream size (log basin area) and water chemistry PC1 for upper and lower basin sites.

Figure 4. Species richness model developed in the upper basin applied to upper basin data.

Figure 5. Application of the upper basin species richness model in the lower basin. Lower basin sites are coded to their level of AMD-impairment and distance from a mainstem river (i.e., Cheat River or Big Sandy Creek).

Figure 6. Ratio of observed and expected species richness from the upper basin species richness model as a function of first water chemistry PCA factor.

Figure 7. Observed commonly occurring species richness as a function of expected common species richness determined from upper basin discriminant function presence/absence models.

Figure 8. Relative cumulative proportion (Y1) and density (Y2) of individuals of commonly occurring species as a function of basin area. The relative cumulative proportion for each species (observed) is compared to the theoretical expected distribution under the null hypothesis that individuals accumulate in proportion to the amount of stream area (m^2) sampled.

Figure 1

Figure 2

Figure 3

Figure 4

Figure 5

Figure 6

Figure 7

Figure 8a

Figure 8b

Figure 8c

Figure 8d

Figure 8e

Figure 8f

Figure 8g

Figure 8h

Figure 8i

Figure 8j

Figure 8k

Figure 8l

Figure 8m

Figure 8n

Figure 8o

Figure 8p

CURRICULUM VITAE

Jason G. Freund

Division of Forestry
West Virginia University
Morgantown, WV 26506-6125
(304) 293-2941 ext. 2432
jfreund@wvu.edu or JasonFreund@adelphia.net

EDUCATIONAL BACKGROUND

Doctorate of Philosophy, West Virginia University (Expected August 2004)

- Dissertation topic: Local and Regional Impairment of Fish Assemblages in a Mined Appalachian Watershed. Advisor: Dr. J. Todd Petty

Master of Science, West Virginia University (2002)

- Theses topic: Seasonal movement and macrohabitat use of largemouth bass (*Micropterus salmoides*) in an Ohio River navigation pool (Belleville Pool). Advisor: Dr. Kyle J. Hartman

Bachelor of Science, University of Wisconsin at Platteville (1995)

- Major: Biology with emphases in Field Biology and Zoology
- Graduated with Honors (GPA = 3.67)

PROFESSIONAL EXPERIENCE

2000 – Present	Graduate Research Associate Division of Forestry, Davis College of Agriculture, Forestry, and Consumer Sciences West Virginia University
1997 – 2000	Graduate Research Associate Division of Forestry, College of Ag., For., & Cons. Sci. West Virginia University
1995 – 1997	Field and Laboratory Technician Department of Agronomy: Dr. Daniel Undersander University of Wisconsin - Madison
1993 – 1995	Genetics Tutor Fish and Amphibian Collection Curator Department of Biology University of Wisconsin - Platteville

HONORS AND AWARDS

Kevin Kinsley Award; Outstanding Graduate Student in Division of Forestry – 2004

- This award was presented by the Division of Forestry at West Virginia University to the outstanding graduate student within the Division.

West Virginia University Student Sub-Chapter of the American Fisheries Society Service Award - 2004

- This award was presented for my efforts with the WV-AFS Student Sub-Chapter. I was one of the founding members of the group, designed and created our webpage (http://www.wvu.edu/~fisheries_society/), presented or arranged talks at meetings, and served on several committees.

Hoyt Fellowship – Fall 2003

- This fellowship, awarded by West Virginia University, replaced my Graduate Research Assistantship for a semester. With this fellowship, I was able to teach an undergraduate Fisheries Management within the Division of Forestry.

TEACHING EXPERIENCE

Methods for Stream Channel Assessment and Analysis: Fundamentals and Principles of Natural Stream Restoration and Design (Level 2)

Co-taught with Steve Kite, Jim Anderson, and Neil Carte

- Course is the second in a series of four that follow Rosgen's Natural Stream Channel Design (NSCD) procedures.
- Course is focused on the assessment of physical processes in stream and their inclusion in NSCD. I developed and presented information regarding habitat sampling and how it can be incorporated into NSCD projects.
- Each instructor led and organized data collection efforts for their groups (10 people per group)

Stream Ecosystem Assessment (WMAN 493)

Co-taught with Dr. J. Todd Petty and George Merovich

- Course focused on the assessment of streams and their watersheds using water chemistry, habitat, and macroinvertebrate and fish assemblages as indicators of stream and watershed condition
- Students learned data collection procedures, quality control and assurance, data analysis, and interpretation.
- Each instructor led and organized data collection efforts for their groups (10 people per group)
- I developed and presented information on Local and Regional influences on Stream Ecosystems, Introduction to Fish Community Assessment, Use and Development of Fish Index of Biotic Integrity (IBI), Physical Habitat Assessment and Rapid Visual Habitat Assessment (RVHA), and Introduction to Statistical

Analysis of Biological Data.

- Field and class work lead to a final paper modeled after Clean Water Act mandated watershed assessments
- For more information: [http://www.forestry.caf.wvu.edu/tPetty/Stream Ecosystem Assessment.html](http://www.forestry.caf.wvu.edu/tPetty/Stream%20Ecosystem%20Assessment.html)

Instructor: Fisheries Management: WMAN 445 – Fall 2003

- The course is a required for all undergraduates in the Wildlife and Fisheries Program at West Virginia University
- The course provides students with an introduction to basic ecological principles as they pertain to natural and impaired watershed. The second half of the course builds upon this foundation and applies basic ecology into the management of fisheries and aquatic resources.
- Taught 43 upper level undergraduate and graduate students
- Created lecture topics, assignments, and exams
- Created and lead all laboratory and field experiences

Guest Lectures and Laboratory Assistance

- Developed and presented several lectures for Introduction to Fisheries Management, Vertebrate Natural History, and Wildlife Ecosystem Ecology
- Assisted with many laboratories and problems sets in Limnology, Vertebrate Natural History, and Introduction to Fisheries Management courses

Teaching Assistant, Ecology and Evolutionary Biology (BIOL 221)

- Teaching assistants (TA) served as advisors for their students that are conducting semester-long experiments
- I developed lectures for laboratories including experimental design, statistical analysis, and basic writing skills
- I served as a “major advisor” for approximately 30 undergraduate students in conducting ecological research
- I was responsible for all grading for the separate 2 credit laboratory experience

Teaching Assistant, General Biology Laboratory (BIOL 104)

- Course was offered to non-biology majors as an option to fill science requirements for graduation
- Developed lectures and quizzes for 4 lab sections
- Students were undergraduate non-biology majors

RESEARCH INTERESTS

- Stream and Large River Ecology
- Community Ecology
- Spatial Ecology
- Ecological Modeling
- Bioassessment Indices
- Local and Regional Determinants of Fish Assemblages
- Population Dynamics and Community Structure

QUANTITATIVE SKILLS

- **Multivariate Statistics:** Application of multivariate statistics to fish ecology and management. Applied experience in PCA, DFA, Cluster Analysis, and Multi-dimensional scaling.
- **Mark-Recapture Population Parameter Estimators:** statistical theory and application
- **Spreadsheets:** age-structured population models; data management
- **SAS:** data manipulation, statistical analysis, model fitting and simulation
- **Habitat Suitability Modeling:** Stream habitat and fish habitat use
- **Study Design:** application of statistical methods to fish ecology and management problems
- **GIS Modeling:** application of geographic information systems and spatial autocorrelation models
- **Multimetric Indices:** applied use and development of multimetric indices
- **Statistics:** applied experience with parametric and non-parametric, univariate and multivariate, and spatial statistics

PUBLISHED PEER-REVIEWED ARTICLES

- Petty, J. T., J. Freund, P. Lamothe, and P. Mazik. 2003. Quantifying instream habitat in the Upper Shavers Fork basin at multiple spatial scales. *Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies* 55:81-94.
- Freund, J. G. and K. J. Hartman. 2002. Influence of depth on detection distance of low-frequency radio transmitters in the Ohio River. *North American Journal of Fisheries Management* 22:1301-1305.

MANUSCRIPTS IN PROGRESS

- Freund, J. G., and J. T. Petty. In Progress. Response of fish and macroinvertebrate bioassessment indices to water quality in an intensively mined Appalachian watershed.
- Freund, J. G., and J. T. Petty. In Progress. Watershed scale disruption of ecological processes in an intensively mined Appalachian Watershed: Fishes as an indicator of watershed impairment. For submission to: *Ecological Applications*.
- Freund, J. G., and J. T. Petty. In Progress. Development of a core/periphery view to assess fish response to watershed scale impairment. For submission to: *Ecological Applications* or *Copeia*.
- Petty, J. T., J. G. Freund, and J. Barker. In Progress. Structures and processes responsible for channel unit formation in a fluvial Appalachian river system. (working title)

Freund, J. G., and K. J. Hartman. In Progress. Importance of Off-Channel Habitats to Largemouth Bass in an Ohio River Navigation Pool. For submission to: Journal of Freshwater Ecology.

Freund, J. G., J. T. Petty, and S. Welsh. In development. Stream size and physical habitat associated with cyprinid hybrids in the Cheat River watershed, West Virginia, with special reference to the Cheat Minnow. For submission to: Copeia.

SELECTED PAPERS PRESENTED AT SCIENTIFIC MEETINGS

2004

Freund, J. G., and J. T. Petty. 2004. "Response of multimetric indices to water chemistry and habitat conditions in the Cheat River Watershed, West Virginia". 134th Annual Meeting of the American Fisheries Society, Madison, WI. (Accepted for Presentation)

2003

Freund, J., and J. T. Petty. 2003. "Local and regional influences on fish communities in the Cheat River Watershed." Mid-Year Meeting of the Southern Division of the American Fisheries Society, Wilmington, NC.

Freund, J. G., and J. T. Petty. 2003. "Fish community response along a disturbance gradient". West Virginia American Fisheries Society, Morgantown, WV.

2002

Freund, J. G., and J. T. Petty. 2002. "Local and regional influences on fish communities in the Cheat River watershed". Spring Meeting of the West Virginia Chapter of the Wildlife Society.

Petty, J. T., J. Freund, P. Lamothe, and P. Mazik. "Combined effects of habitat and water temperature on brook trout distributions: implications for habitat restoration". 132nd Annual Meeting of the American Fisheries Society.

2001

Petty, J. T., J. Freund, P. Lamothe and P. M. Mazik. 2001. "Quantifying microhabitat characteristics of hydraulic channel units in the upper Shavers Fork basin". Annual Meeting of the Southeastern Association of Fish and Wildlife Agencies, Louisville, KY. (Presented by: Jason Freund)

Petty, J. T., P. Lamothe, J. Hansbarger, and J. Freund. "Quantifying spatial variation in trout habitat suitability at a watershed scale". 47th Annual Tri-State Fisheries Conference, Huntington, WV. (Presented by: Todd Petty)

2000

Freund, J. G., and K. J. Hartman. "Influence of conductivity and depth on low-frequency underwater radio telemetry signal attenuation". 56th Annual Northeast Fish and Wildlife Conference, Charleston, WV.

1999

Freund, J. G. and K. J. Hartman. 1999. "Evaluation of a pilot largemouth bass stocking program using radio telemetry". 1999 Mid-Year Meeting of the Southern Division of the American Fisheries Society, Chattanooga, TN.

Freund, J. G. and K. J. Hartman. 1999. "Overwintering habitat selection by largemouth bass (*Micropterus salmoides*) in the Ohio River utilizing radio telemetry". 1999 Tri-State American Fisheries Society Meeting. Ashland KY.

1998

Freund, J. G. and K. J. Hartman. 1998. "Proposed Research: Largemouth bass movement and habitat use in the Ohio River". 1999 Mid-Year Meeting of the Southern Division of the American Fisheries Society, Chattanooga, TN.

PRESENTATIONS AT SERVICE ORGANIZATION MEETINGS

- Presented Ph. D. research at West Virginia University Student Sub-Chapter of AFS
- Tour Leader: Canaan Valley National Wildlife Refuge, WV – Led tour of aquatic resources on the CVNWR
- Presented information at Northern Kentucky Fly Fishers (Hebron, KY)
- Presented information at Kanawha Valley Chapter of Trout Unlimited (Charleston, WV)
- Presented information at Almost Heaven Chapter of Trout Unlimited (Beckley, WV)
- Presented Ph. D. research at P. Pendleton Kennedy Chapter of Trout Unlimited (Morgantown, WV)
- Presented Master's research findings at the West Virginia Bass Federation (Parkersburg, WV)

PROFESSIONAL SOCIETIES AND SERVICE

- American Fisheries Society (AFS)
- Ecological Society of America (ESA)
- West Virginia University Wildlife and Fisheries Society; President (1998)
- Southern Division American Fisheries Society; Committee Member, Student Affairs (2000-2001)

- West Virginia University American Fisheries Society Student Sub-Chapter; Founding member (2000), Secretary (2001-2002); Web Site Developer; Student Colloquium Committee Member
- Moderator; 2003 Mid-Year Meeting of the Southern Division of the AFS, Wilmington, NC and 134th Annual Meeting of the American Fisheries Society, Madison, WI. (Upcoming event)

EDITORIAL DUTIES

Asked to referee articles for the following journals:

North American Journal of Fisheries Management

Southeastern Association of Fish and Wildlife Agencies

Transactions of the American Fisheries Society

RELEVANT TRAINING, EXPERIENCES, AND SERVICE

- American Red Cross First Aid and CPR
- American Red Cross Community Water Safety
- U. S. Department of the Interior Motor Boat Operators Certification
- Maryland Motor Boat Operator's Certification
- Small Stream Assessment (Pennsylvania State University)
- Trout Unlimited; Founding member of West Virginia University Sub-Chapter, Committee member of P. Pendleton Kennedy Chapter (Morgantown, WV)
- Assistance with graduate research projects; amphibian surveys, Wetland/Riparian nesting birds, large river, and small stream research projects

REFERENCES

Dr. J. Todd Petty

West Virginia University
Division of Forestry
Morgantown WV 26506-6125
(304) 293-2941 ext. 2417
jtpetty@wvu.edu

Dr. Patricia Mazik

Cooperative Fish and Wildlife Research Unit
West Virginia University
Morgantown WV 26506-6125
(304) 293-2941 ext. 2431
pmazik@wvu.edu

Dr. Stuart Welsh

Cooperative Fish and Wildlife Research Unit
West Virginia University
Morgantown WV 26506-6125
(304) 293-2941 ext. 2419
swelsh@wvu.edu

Dr. Kyle J. Hartman

West Virginia University
Division of Forestry
Morgantown WV 26506-6125
(304) 293-2941 ext. 2494
Hartman@wvu.edu

Dr. Michael P. Strager

West Virginia University
Department of Resource Management
2006 Agricultural Sciences Building
PO Box 6108
Morgantown WV, 26506
(304) 293-6253 x4453
mstrager@wvu.edu

Jennifer Barker Fulton

Research Associate
West Virginia University
Division of Forestry
Morgantown WV 26506-6125
(304) 293-2941 ext. 2321
jebarker@mail.wvu.edu